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COOPERATIVE AGREEMENT NUMBER DAMD17-98-2-8006

TITLE: Scientific Coordination and Adaptive Management and Experimental Restoration of Longleaf Pine Community Structure, Function, and Composition

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REPORT DATE: March 1999

TYPE OF REPORT: Annual

PREPARED FOR: U.S. Army Medical Research and Materiel Command
Fort Detrick, Maryland 21702-5012

REPORT DOCUMENTATION PAGE

Form Approved
OMB No. 0704-0188

1. AGENCY USE ONLY *(Leave blank)*	2. REPORT DATE	3. REPORT TYPE AND DATES COVERED
	March 1999	Annual (31 Mar 98 - 30 Mar 99)

4. TITLE AND SUBTITLE
Scientific Coordination and Adaptive Management and Experimental Restoration of Longleaf Pine Community Structure, Function, and Composition

5. FUNDING NUMBERS
DAMD17-98-2-8006

6. AUTHOR(S)
Hardesty, Jeffrey L.

7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES)
Nature Conservancy
Arlington, Virginia 22209

8. PERFORMING ORGANIZATION REPORT NUMBER

9. SPONSORING / MONITORING AGENCY NAME(S) AND ADDRESS(ES)
U.S. Army Medical Research and Materiel Command
Fort Detrick, Maryland 21702-5012

10. SPONSORING / MONITORING AGENCY REPORT NUMBER

19990511 080

11. SUPPLEMENTARY NOTES

12a. DISTRIBUTION / AVAILABILITY STATEMENT
Approved for public release; distribution unlimited

12b. DISTRIBUTION CODE

13. ABSTRACT *(Maximum 200 words)*

This report addresses aspects of sandhill restoration in fire-suppressed, longleaf pine-dominated sandhills on Eglin Air Force Base, Florida. Responses of selected variables to three hardwood reduction techniques (growing season fire, application of hexazinone herbicide, and chainsaw felling/girdling) were compared to control on two measures of community similarity between restoration and reference plots. The selected response variables included densities of longleaf pine (*Pinus palustris*) and oaks (*Quercus* spp.) of different diameter classes, and variables for soil chemistry, tree species, groundcover vegetation, soil/litter arthropod density, herb-layer arthropod density, and breeding bird species. We detected significant effects of restoration treatments on the similarity between restoration and reference plots for 7 fall variables and 6 spring variables.

In a separate study, we measured the effects of Choctawhatchee sand pine (*Pinus clausa* var. *immuginata*) removal in sand pine-invaded sandhills. We documented the impact of sand pine removal on groundcover plant species, contrasted results from removal plots to reference sites, tested colonization of sand pine seedlings, and measured longleaf pine seedling survival. Plant species and life form densities increased post harvest; regeneration of sand pine seedlings decreased with distance from plot corners; and survival of planted longleaf pine seedlings was approximately 80%.

14. SUBJECT TERMS	15. NUMBER OF PAGES
arthropods; birds; community similarity; ecological restoration; fire; hexazinone; longleaf pine; oaks; sandhills; sand pine; seedlings; soil chemistry; tree felling and girdling; vegetation	88
	16. PRICE CODE

17. SECURITY CLASSIFICATION OF REPORT	18. SECURITY CLASSIFICATION OF THIS PAGE	19. SECURITY CLASSIFICATION OF ABSTRACT	20. LIMITATION OF ABSTRACT
Unclassified	Unclassified	Unclassified	Unlimited

FOREWORD

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PREFACE

This report is intended for the staff of the Natural Resources Division at Eglin Air Force Base. We direct our style to those with technical expertise in forest, wildlife, and fire management. We have attempted to accommodate the majority of those interested in this report.

We have written this report in chapters that focus on specific issues. This format allows the reader to concentrate on a topic of interest without having to read the entire report. The format has resulted in repetition of certain basic information, figures, and references in each chapter. We beg the cover-to-cover reader's indulgence on this point, if any souls are so determined!

This report departs from our past efforts in two important aspects. First, we limit ourselves to three core chapters whose contents are strictly management oriented. Chapters that focus only on the responses of a cohesive group of species and variables to restoration treatments are deliberately avoided unless managers showed a strong desire to see such analyses. The chapter on the effects of hardwood reduction techniques on tree densities and recruitment is one such exception. On the other hand, a synthetic chapter that directly measures the restoration success of hardwood reduction techniques across all taxa and variable groups is the centerpiece of the report. Second, we reduce the presentation of quantitative data and statistics. This report, therefore, does not contain extensive statistical analyses and appendices, although this information is available by request.

The body of the report contains the following sections:

1) Executive Summary;
2) Introduction – Provides a brief background on important themes in restoration ecology and their application to the ecology of the longleaf pine ecosystem, and explains the structure of the report;
3) Longleaf pines and oaks – Presents the methods, results, discussion, and management implications of post-treatment effects on longleaf pine and oak diameter classes and tree recruitment;
4) Restoration success – Presents the methods, results, and discussion of the effects of restoration techniques on the similarity between restored and reference plots for multiple taxa and variables;
5) Sand pine removal – Presents the methods, results, and discussion of the impact of sand pine removal on groundcover vegetation.

ACKNOWLEDGMENTS

The Longleaf Pine Restoration Project acknowledges the assistance and funding of the Natural Resources Division of Eglin Air Force Base (EAFB) in setting up the project. We are especially grateful to Richard McWhite for his continued support of the research. We thank Lou Ballard, James Furman, Carl Petrick, and Steve Seiber for their administrative, technical, and field support. We are truly grateful to all staff who participated in the protection of our plots from wildfires and who saw to the repairs of our office's internal plumbing. We thank the Range Operation Control Center (ROCC) for keeping us safe.

We are very grateful to Florida Natural Areas Inventory (FNAI) for providing natural community and rare plant and animal element occurrence records. Especially thanked is Carolyn Kindell (FNAI) for sharing her extensive ecological knowledge of the natural communities at EAFB. Dr. Loran Anderson (Florida State University) and Dr. David Hall (Gainesville, FL) helped resolve many taxonomic problems with grasses and forbs. We thank Dr. Anderson, Kent Perkins (University of Florida), and Dr. James Burkhalter (University of West Florida) for the use of their collections and library resources.

Dr. R. Wills Flowers (Florida A&M University) has continued to cheerfully volunteer much advice and expertise from leaf beetle identifications to fixing the leaky sink. The morphospecies analysis would not have been possible without the identifications performed by the following specialists: Dr. Jeffrey M. Cumming (Agriculture & Agri-Food Canada); Dr. Mark Deyrup (Archbold Biological Station); Dr. R. Wills Flowers (Florida A&M University); Dr. Susan Halbert (Florida State Collection of Arthropods); Dr. Richard J. Snider (Michigan State University); Dr. Lionel Stange (Florida State Collection of Arthropods); Dr. Gary Steck (Florida State Collection of Arthropods); and Dr. Michael C. Thomas (Florida State Collection of Arthropods). We have enjoyed the many hours of phone conversations and e-mail with Drs. Deyrup, Flowers, and Snider about the habits of the fascinating insects they study, and we appreciate their insights.

We thank the staff of Tall Timbers Research Station for their assistance and support, especially Anne Bruce for locating literature, Kay Gainey for providing helpful plant literature, and Angus Gholson for assistance with common plant names. We thank Dr. Franklin Percival and Barbara Fesler from the National Biological Service at the University of Florida for their administrative support.

We thank Jora Young from The Nature Conservancy/Florida Regional Office (TNC/FLRO) for her continued support and TNC/FLRO for financial commitment during budgetary transitions. We are truly grateful to Suzanne Woodward from the Public Lands Program (TNC/Gainesville) for all her administrative help. Without her and her predecessor, Susan Feller, we would be in sorry shape. Raymond Moranz, Jodi Slapcinsky, Jill Fisher and many volunteers (TNC/Gainesville) provided much help by locating and photocopying references. We feel honored to work with these people.

We gratefully acknowledge Eric Anderson, Marc Barb, Keith Blandford, Tina Bastien, Kevin Campbell, Dean Demarest, Andrea Iosue, Elaine Levine, Molly Northrup, H. LeRoy Rodgers, Diana Swan, and LeAnn West for data entry or collecting a significant part of the 1996 data used in this report. Special thanks to H. LeRoy Rodgers for miracles performed from 1994 to 1997.

We thank those who reviewed all or part of an earlier draft of this report: Dr. Dale Brockway, Dr. James Hanula, Dr. Jeffrey Hyman, Dr. Frances James, Carolyn Kindell, Dr. Katherine Kirkman, Dr. John Kush, and Dr. Deanna McKay. Any shortcomings remaining are, of course, the fault of the authors.

1. EXECUTIVE SUMMARY

Restoration of fire-suppressed longleaf pine (*Pinus palustris*) sandhill communities requires reduction of the stature or removal of encroached hardwoods and off-site fire-sensitive pines (*Pinus* spp.) to stimulate the recovery of other understory components. We compared the effects of three hardwood reduction techniques (growing season fire, application of the ULW® form of the herbicide hexazinone, chainsaw felling/girdling) to a no-treatment control on selected variables in fire-suppressed longleaf pine-dominated sandhills at Eglin Air Force Base (EAFB), Florida. The selected response variables included densities of longleaf pine and oak (*Quercus* spp.) of different diameter classes and variables for soil chemistry, tree species, groundcover vegetation, soil/litter arthropod density, herb-layer arthropod density, and breeding bird species. Initial treatments were applied in the spring and summer of 1995. Felling/girdling and ULW® plots were also burned for fuel reduction from early March to late April in 1997. Treatment plots were 81 ha (200 acres) in size, each with six replicates distributed among six blocks of a randomized split-plot complete block design. The split-plot level of the design was not presented in this report. Whenever possible, restoration effectiveness was also compared to a restoration "model" derived from six 81-ha (200-acre) frequently-burned reference sites. Pre- and post-treatment data, respectively, were collected from spring 1994 to spring 1995 and from spring 1995 to fall 1998.

In a separate study, we measured the effect of the Choctawhatchee variety of sand pine (*Pinus clausa* var. *immuginata*) removal on the groundcover in sandhills where this species had become dominant in the overstory. The goals of this study were: a) to document the impact of sand pine removal on groundcover plant species richness and densities for 4 years after removal; b) to contrast results from removal plots to those from frequently-burned longleaf pine-dominated reference sites; c) to test whether sand pine seedlings colonized more rapidly in the periphery than in the core of the plots following removal of trees; and d) to measure the survival of containerized longleaf pine seedlings for 3 years after planting. Sand pine removal was replicated with six 81-ha (200-acre) plots. Pre- and post-removal data, respectively, were collected in the fall 1994 and from fall 1995 to 1998.

Effects of Hardwood Reduction Techniques on Oaks and Longleaf Pines. Pre-treatment size distributions of oaks followed a negative-exponential curve in all treatments. The size distribution in reference plots was practically flat with low tree densities in all diameter classes. Oak densities were significantly reduced in the ULW® and felling/girdling plots in 1995. Growing season fire resulted in significantly more hardwood topkill than in the control plots for trees in the 0-4.9 cm and 15-19.9 cm classes. In both classes, topkill was less than 20%. Delayed mortality of approximately 50% among smaller oaks was observed in burn plots in 1997. In both 1995 and 1996, the greatest reduction of oak juvenile density (<1.4 m high) was caused by ULW® application. Control plots contained significantly fewer oak juveniles than the burn and felling/girdling plots. Reference plots contained the lowest and most variable oak juvenile densities.

Size distributions of longleaf pine across all plots were bimodal with modes at 0-4.9 cm and 25-29.9 cm. The highest mode was at 0-4.9 cm in treatment plots and at 25-29.9 cm in reference plots. Only fire quantitatively changed the distributions by the attrition of the

smallest trees >1.4 m high in 1995 and 1997. Fire caused approximately 50% decreases in longleaf pine juvenile (<1.4 m high) density in 1995 and 1997. Longleaf pine juveniles were unaffected by other treatments. Juvenile densities were slightly higher and substantially more variable in reference plots than in treatments. Treatments had no effect on recently germinated longleaf pine seedlings. Densities ranged between 75 (control) and 105 seedlings/20 m^2 (felling/girdling) in treatment plots (15,182 and 21,255 seedlings/acre, respectively). Median reference plot densities were 205 seedlings/20 m^2 (41,498 seedlings/acre). In 1997, fuel reduction burns decreased densities to <5 seedlings/20 m^2 (<1,012 seedlings/acre) in ULW® and felling/girdling plots. Seedling densities dropped by approximately 50% in control and burn plots, although these sites received no manipulations after 1995. Seedling densities only decreased by 22% in reference plots, which did experience some fires.

Effects of Hardwood Reduction on Restoration Success. Restoration success for a group of variables was defined by an increasing similarity between treatment plots and reference sites. For fall measurements taken in 1995-1997, we detected significant effects of restoration treatments on the similarity between restoration and reference plots in 7 of 27 groups of variables: basal area of all tree species in 1995 and 1997, groundcover plant density in 1997, plant life form flower density in 1995, vegetation cover groups in 1996 and 1997, and arthropod species/morphospecies density in 1996. For the spring measurements taken in 1996-1998, treatment effects on similarity were detected in 6 of 12 groups of variables: arthropod species/morphospecies densities in 1997, breeding bird detection rates from 1996 to 1998, and the number of breeding bird species (one variable group) in 1996 and 1997. Restoration treatments were not significantly more similar to the reference condition for soil chemistry, tree density, groundcover life form density, number of plant species, or herb-layer and soil/litter arthropod family density in any season or year. Growing season burning increased the similarity between restoration and reference plots for arthropod species/morphospecies density and groundcover plant density. The similarity of groundcover plant density to reference sites decreased in felling/girdling and ULW® plots subjected to fuel reduction fires. ULW® application decreased the similarity to the reference condition for the density of flowers of plant life forms. In 1995 and 1997, fire increased percent cover group similarity by burning fine fuels. Treatments increased the similarity to the reference condition for breeding bird detection rates and species richness.

Management Implications of Hardwood Reduction Techniques. Identifying the most ecologically effective method for reducing hardwoods depends on the time frame within which management objectives specify restoration of the system from a fire-suppressed to a maintenance condition. If time is not the highest priority, our results show that growing season burning is the most cost-effective and ecologically beneficial method. On the other hand, if immediate results are necessary, mechanical or chemical reduction of the midstory provides an effective but more expensive solution to removing hardwoods. These techniques, however, do not imply that other groundcover components will be restored. Because the costs of felling operations and ULW® application are comparable, the choice between these options should be based on maintenance of hardwood resprouts and the side effects of herbicides. If resprouting of hardwoods is unacceptable even under fire management, herbicide may be needed. We would not recommend ULW® as the herbicide of choice because we have shown that it negatively impacted groundcover plant and

arthropod variables, as well as in relation to reference sites. Brushbullet version of hexazinone (Pronone®) or herbicide injection by hypo-hatchet may reduce impacts to the understory communities, but we have not investigated these options here. If high densities of resprouts are acceptable to managers during fire intervals, felling/girdling should be less detrimental to the understory and cheaper to apply than brushbullet or injecting herbicide. Moreover, bird and arthropods appear to benefit from oak resprouts. The greatest danger to longleaf pines and understory components from felling/girdling is the intensity of fuel reduction burns applied two to three years later. These burns can be more intense in felling/girdling than in ULW® plots because of heavy fuels.

Sand Pine Removal. The number of plant species decreased from 30 to 26/400 m^2 from pre- to the first sand pine post-harvest year, but then increased beyond pre-removal levels to 46 species/400m^2 from 1995 to 1997, followed by a small reduction in 1998. The number of plant species was greater in reference plots compared to removal plots prior to 1997 only. New plant species in removal plots were predominantly ruderal and, therefore, not representative of the reference condition. Clonal shrubs and large trees were the only groundcover life forms whose densities continued to decline following sand pine harvest. Groundcover graminoids, non-legume forbs, legumes, non-clonal shrubs, woody vines, and small trees showed increasing densities after harvest. Except for densities of non-clonal shrubs and woody vines, the densities of other life forms were greater in reference plots than in removal plots. Percent cover of graminoids, forbs, woody species, woody litter, and bare ground increased after an initial decrease following harvest. Predictably, fine litter cover declined erratically with time. Graminoid and forb cover increases closely matched those of graminoid and forb densities and were within the range of reference plot cover values. Two to four years post-harvest, a gradient in density of regenerating sand pine seedlings was apparent, decreasing with the distance from the corner of the plots. On average, 80% of planted longleaf pine seedlings survived over the first two years in removal plots. Total longleaf pine seedling densities increased by as much as one order of magnitude in a few plots in 1998, but by then we had stopped differentiating between planted and naturally regenerated seedlings. Overall, silvicultural removal of sand pine with minimal soil disturbance followed by fire and planting of longleaf pine appears to be a promising restoration approach in sandhills invaded by sand pine. However, continued management to monitor and eradicate invasive non-indigenous species is needed.

2. INTRODUCTION

The National Research Council's (1992) definition of ecological restoration is "returning a system to a close approximation of its condition prior to disturbance, with both the structure and function of the system recreated." As pointed out by Palmer et al. (1997), this definition implies that the ecological endpoint (target of restoration) and the measures to track change from a degraded state to a system with high integrity (reference sites) are known. Both assumptions are notoriously difficult to satisfy.

Reference sites may be locally and regionally lost, described only in qualitative historical notes, or so fragmented or reduced in extent that demographic isolation makes quantitative assessment questionable for some taxa. Equally important is the fact that the components of reference systems change (DeAngelis and Waterhouse 1987), sometimes rapidly, such that there is an obvious need to re-quantify endpoint conditions as often as restored systems are sampled. Therefore, when restoration studies rely on historical notes or on descriptions of regional but distant sites, the dynamics of reference systems cannot be addressed. It is a luxury to have access to local quantifiable reference systems.

But assuming such luxury, reducing the choice of metrics from potentially hundreds or thousands of variables to a few metrics that will be sensitive enough to track ecological change following restoration is a major challenge in restoration ecology (e.g., Karr 1991, Barbour et al. 1996). Restoration ecologists can borrow from various conceptual and quantitative methods in community ecology to extract meaning from complex sets of variables (Palmer et al. 1997). Ideally, several methods of analysis should be compared. Researchers can decide to measure restoration success from a small subset of species because they were introduced to the site, they are threatened and, therefore, deserve special consideration, or they are species that greatly affect the structural and functional properties of the ecosystem and create habitats for other species. Alternatively, species can be partly ignored so that a small group of geophysical variables can be tracked to document functional integrity. The above options all share the premise that a subset of variables accounts indirectly for a large fraction of the total system variation. This hypothesis has been the subject of a long-standing debate in ecology (Holling 1992, Huston 1994). If the whole system is not driven by key components or the difficulties of identifying key components are logistically overwhelming (Palmer et al. 1997), then reducing the larger ecosystem into manageable units is the next step.

Classic examples of manageable ecological units are life forms for vegetation and functional groups for plants and animals. The advantage provided by these groupings is that nearly all species of plants and animals at a site can be sorted into a small number of categories. Two drawbacks are that the definition of groups can be arbitrarily manipulated and opposite contributions by species may dilute the response of the group when used to measure the effect of management techniques.

The first drawback can be alleviated somewhat by adhering to independent life form classification keys (e.g., Mueller-Dømbøis and Ellenberg 1974) and observing of common foraging strategies for animals (e.g., detritivores, herbivores, insectivores, non-parasitoid predators, parasitoids, and below-ground predators) (e.g., Root 1967). When foraging information is lacking for some species, as is the case for many arthropods, groupings should

remain broadly defined. When foraging information is generally lacking for the taxa, functional groups may not be applicable, and studying the common species may be the only alternative. Also difficult to categorize are species with fundamentally different foraging habits associated with multiple life stages (e.g., the insect larva is predatory, but the adult is a nectar feeder). Because these animals are not omnivores, one can sort species by the life stage most related to management objectives or split life stages into separate categories.

The second drawback generally cannot be avoided, but may be alleviated by novel quantitative methods if needed. For example, one may want to weigh each species' abundance by its relative abundance. This is similar to financial market indices, in which companies are weighted by capitalization (e.g., Standard and Poor's 500). Also, species can be weighted by a "quality" weight within the plant community as in the Floristic Quality Index (Burke and Nol 1998) or by a statistical coefficient that reflects the percent spatial variation a particular species accounts for in the whole community. We are not aware of any such efforts in ecology, although the index of biotic integrity proposed by Karr (1991) somewhat approximates this statistical approach.

Quantitative methods that group variables according to their variation across different levels of presumed ecological factors (ordination techniques) are sometimes the best tools to differentiate the less responsive variables from the sensitive ones. Ordination techniques are used to sort variables into statistical groups that have simple ecological meaning. An innovative and promising multivariate method beyond traditional ordination has recently been proposed to identify indicator species (Dufrêne and Legendre 1997). This computational and statistical method seeks variables that maximally associate with a group of ecologically similar sites. Although powerful for handling large multivariate data sets, these techniques make the assumption that those variables that account for the largest fractions of the total variation before management has been applied to a system will be affected most by management activities during restoration. For example, graminoid density may be singled out by ordination as the best indicator of fire frequency, because its value differs greatly between frequently-burned reference sites and fire-suppressed sites. During restoration by fire, however, researchers may find that graminoid density does not respond significantly to fire compared to fire-suppressed control plots. Other variables not retained by ordination (e.g., forb density) may have significantly changed and approached the value observed in the reference sites. These scenarios are unpredictable, but not rare (e.g., Boyer and Zedler 1996, Hardesty, Gordon, et al. 1997, Provencher et al. 1997, 1998).

Identifying metrics for ecological restoration using experimental designs has several advantages over the strictly correlational approach of ordination. Parametric statistics and hypothesis testing can be used to find sufficiently sensitive metrics among the hundreds of possible variables. As the number of variables tested increases, however, special efforts should be taken to maintain the significance probability at a chosen level and the independence of tests when variables are correlated. The value of this method is that the variation of a metric subject to manipulation can be compared to a no-treatment control. Moreover, controlled studies should be able to account for initial conditions that could possibly confound perturbation effects, and replication of manipulations allows estimates of variability within treatments. One limitation to fully measuring restoration success and identifying metrics of ecological condition is that the restoration techniques cannot be statistically compared to the reference condition because it is not part of the experimental

design. The reference condition is not a replicable treatment and it cannot be randomized in space. Moreover, reference plots are unlikely to be random samples of the reference condition. This limitation is important because metrics and success only make sense with respect to the variation observed in reference condition (Palmer et al. 1997). One of the central goals of this report is to statistically overcome this limitation while using the results of a restoration project in fire-suppressed, longleaf pine (*Pinus palustris*)-dominated sandhills at Eglin Air Force Base (EAFB), Florida.

EAFB may have the largest remaining longleaf pine holdings in public ownership (≥144,000 ha or 360,000 acres) (Outcalt and Sheffield 1996), of which old-growth forests account for at least 770 ha (1,928 acres) (Provencher et al. 1996, Kindell et al. 1997). This extent is of conservation importance because <2% of the longleaf pine landscape remains (Landers et al. 1995), and these forests harbor some of the highest numbers of species in North America, including many that are rare and endemic (Walker and Peet 1983, Hardin and White 1989, Engstrom 1993, Walker 1993). This forest community was once dominant in the coastal plain from Virginia to Texas along the Atlantic and Gulf coasts and extended northward into Alabama and Mississippi (Frost 1993, Landers et al. 1995, Plunkett and Hall 1995).

EAFB has been under the management of the U.S. Department of Defense (DoD) since 1930, when it was acquired from the Choctawhatchee National Forest and private inholdings. Nearly all of its acreage has been degraded by decades of fire suppression or unnatural fire regimes, logging, turpentining, overgrazing, and other land management practices. Soil alterations from forestry or military activities and an extensive network of roads are common anthropogenic disturbances on this landscape. In addition, a significant degree of habitat alteration (e.g., groundcover suppression) has occurred from colonization by the Choctawhatchee variety of sand pine (*P. clausa* var. *immuginata*) in fire-suppressed and soil-altered longleaf pine areas. This encroachment is especially intense along the coastal lowlands where nearby populations of sand pine occur naturally.

Although much of EAFB's sandhills has been degraded to various degrees (>100,000 ha), there are several stands where, by virtue of frequent fire from nearby bombing ranges and limited historical logging, the hardwood midstory is greatly reduced, a dense and diverse groundcover has been maintained, and many old growth longleaf pines remain. Most of EAFB's longleaf pine old growth and degraded sandhills have understories that are dominated by bluestem grasses (*Andropogon* spp. and *Schizachyrium* spp.) and low panic grasses (*Dichanthelium* spp.), and wiregrass (*Aristida beyrichiana*) is practically absent from the western and northern halves of the Base (Rodgers and Provencher 1999). Provencher et al. (1998) also showed that groundcover woody vegetation (<1.4 m tall) is more abundant than herbaceous vegetation at EAFB. The present condition of these sites is consistent with historical accounts of relatively unaltered sandhills as vast landscapes of monotypic, open canopies of longleaf pine with groundcovers containing rich assemblages of graminoids and forbs (Bartram 1791, Flint 1826, also reviewed in Schwartz 1994).

The system states and restoration challenges that EAFB managers face have been expressed in a simple conceptual model of sandhill ecosystem degradation and recovery (Fig. 2.1; Provencher et al. 1997). The overall model "space" captured the observed dynamic range of possible sandhill states or starting conditions (as expressed by alternative community structure and composition of forest patches or stands) observed on EAFB over

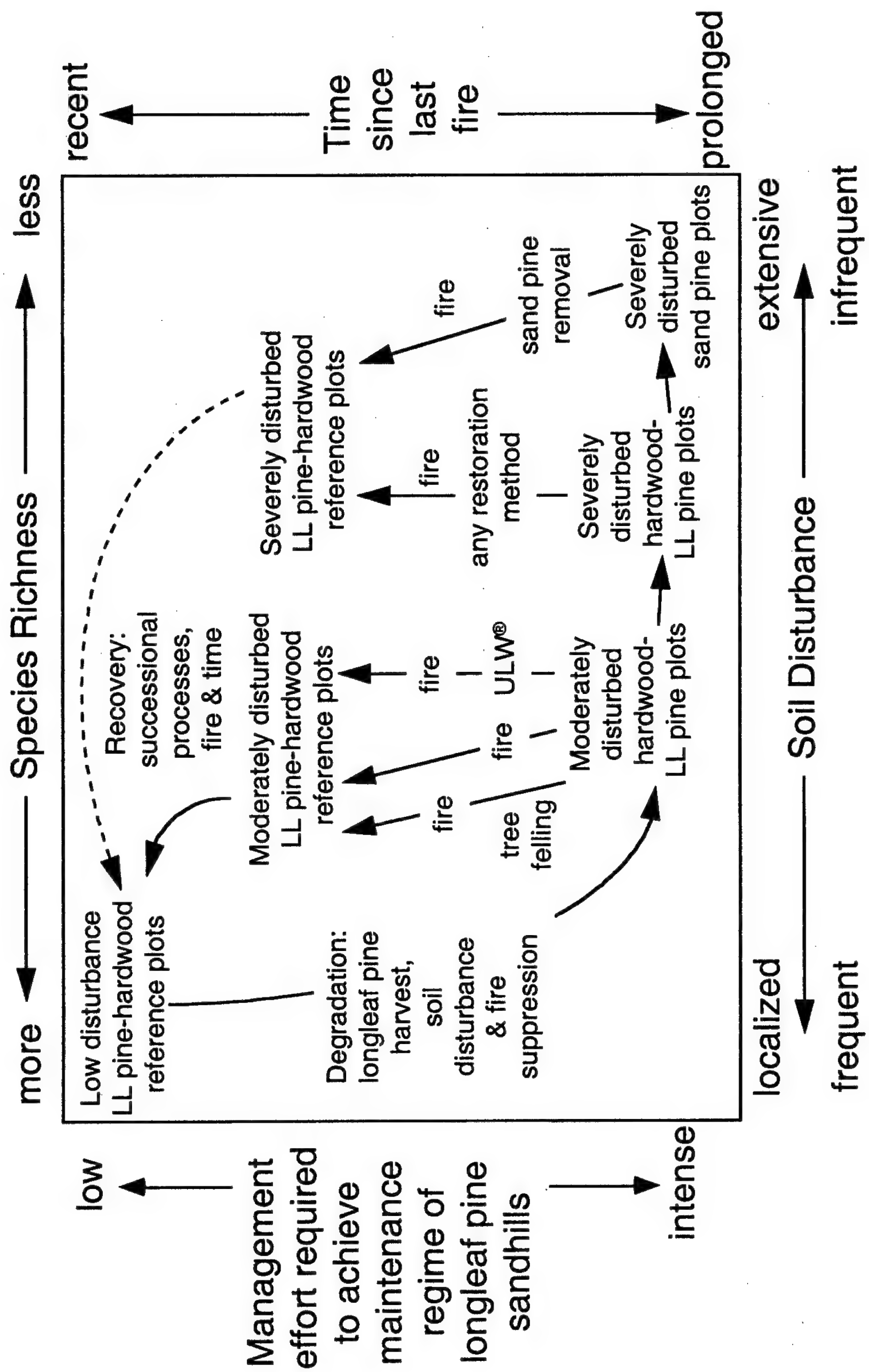

Fig. 2.1. Model of longleaf pine sandhill disturbance and recovery. Arrows indicate processes; their lengths are not proportional to time. See Introduction for explanation. Abbreviation: LL pine = longleaf pine.

time. These different starting conditions were classified primarily on the basis of the presence of old growth longleaf pine individuals (>150 years old), the presence of a co-dominant mid- or overstory of hardwoods and/or sand pine, and the dominance of the understory by only a few, primarily ruderal (i.e., weedy), plant species. These states were also described in a cellular automata model of fire effects on the dynamics of longleaf pine forests at EAFB (Peterson et al. 1998). The different starting conditions are known or hypothesized to be the result of past anthropogenic soil disturbance and fire suppression. Similarly, each of these different sandhill patches are assumed to have different trajectories across model space and time depending on the starting conditions, management intervention type, and biological recovery processes (e.g., plant succession, nutrient flux).

Previous reports (Provencher et al. 1996, 1997) summarized the starting conditions of "restoration" plots and "reference" plots in both the hardwood removal experiment and the sand pine removal study (Fig. 2.1). Post-treatment reports (Provencher et al. 1997, 1998, and this report) focused on the response of biological variables to treatments (as depicted in the lower middle and right of the model). The trajectories in the model depict the relative differences in the generalized biological response to the various treatments (Fig. 2.1). Empirical evidence and experience suggest that if left untreated and in the absence of fire, each of the different starting conditions will move down and, over time, to the lower right of the model (i.e., degraded state). If treated, each model state is assumed to follow a unique trajectory, depending on the degree of soil disturbance and the season and frequency of fire. Soil disturbance can be caused by a variety of factors, such as animals excavating burrows (local and somewhat frequent) or heavy machinery churning soil during timber harvesting (extensive and infrequent). A chronosequence study on EAFB suggested that the time of recovery for a severely soil disturbed and fire suppressed plot may be >50 years (Provencher et al. 1996). In this study, we compared understory vegetation cover among three 20.2-ha (50-acre) plots that were estimated to have been thinned in the 1990s, 1970s, and 1930s. Similarity measures showed that the most recently thinned plot least resembled any other plots, whereas plots with longer recovery periods were more similar to the 1930s plot. Our most recent report (Provencher et al. 1998) provided preliminary evidence in support of the general model in Fig. 2.1 and suggested both trajectories for certain aspects of the sandhill community as well as potential indicators of biological response. We continue to validate the model in this report.

In fire-suppressed, longleaf pine-dominated sandhills at EAFB, we experimentally compared the effects of three hardwood reduction techniques (growing season burn, ULW® form of the herbicide hexazinone, and mid-story mechanical felling/girdling) and a no-treatment control on: a) the densities of longleaf pine and oak from different diameter classes and of immature longleaf pines and oaks <1.4 m in height (Chapter 3); and b) the similarity between restoration and reference plots for soil chemistry, tree species, groundcover vegetation, soil/litter and herb-layer arthropod density, and breeding bird species (Chapter 4). We measured the response of plant species densities and richness to sand pine removal in sandhills where this species had replaced the original longleaf pine in the overstory (Chapter 5). A timeline of our project is summarized in Fig. 2.2 to help the reader sort out the complex nature of our sampling and data.

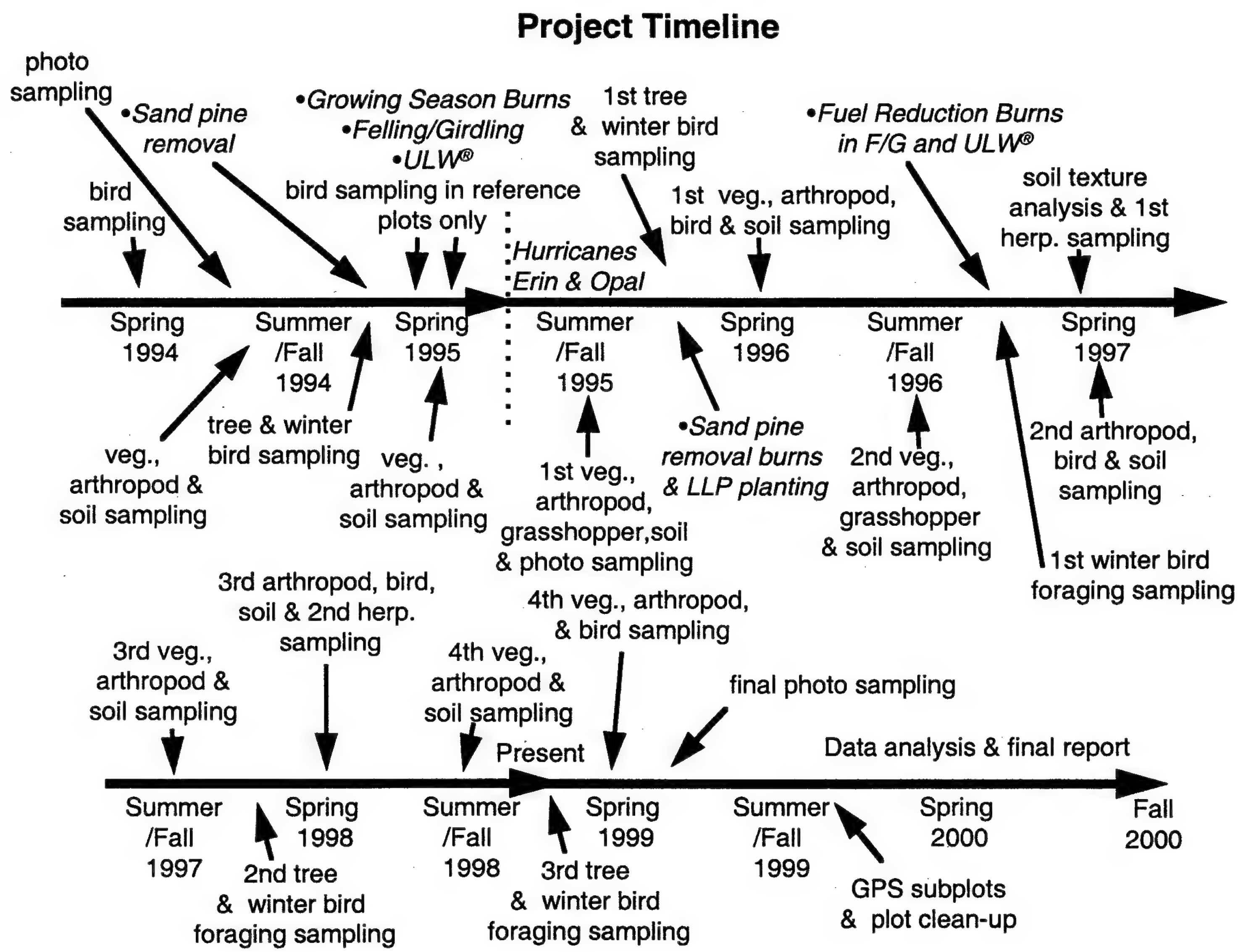

Fig. 2.2. Timeline of the Longleaf Pine Restoration Project. Management activities and hurricanes are printed in italics. The dotted line indicates the end of pre-treatment sampling. Abbreviation: LLP = longleaf pine.

3. EFFECT OF HARDWOOD REDUCTION TECHNIQUES ON LONGLEAF PINES AND OAKS IN FIRE-SUPPRESSED SANDHILLS AT EGLIN AIR FORCE BASE, FLORIDA

ABSTRACT

Restoring fire-suppressed longleaf pine (*Pinus palustris*) sandhill communities often includes reducing hardwood structure before reestablishing maintenance fire regimes. Using a randomized complete block design, we compared the initial effects of three hardwood reduction techniques (growing season burning, application of the ULW® form of the herbicide hexazinone, and midstory chainsaw felling/girdling) and a no-treatment control on hardwood and longleaf pine densities in fire-suppressed sandhills at Eglin Air Force Base, Florida, USA. Treatments were applied in the spring and summer of 1995. Felling/girdling and ULW® plots were also burned for fuel reduction from early March to late April in 1997. Pre-treatment diameter distributions of oaks followed a negative-exponential curve in all treatments. The size distribution in reference plots was practically flat with low tree densities in all diameter classes. Oak densities were significantly reduced in the ULW® and felling/girdling plots in 1995. Growing season fire resulted in significantly more hardwood topkill than the control plots for trees in the 0-4.9 cm and 15-19.9 cm classes in 1995. In both classes, topkill was less than 20%. Delayed mortality of approximately 50% among smaller oaks was observed in burn plots in 1997. In both 1995 and 1996, the greatest reduction of oak juvenile density (<1.4 m high) was caused by ULW® application. Control plots contained significantly fewer oak juveniles than the burn and felling/girdling plots. Reference plots contained the lowest and most variable oak juvenile densities. Size distributions of longleaf pine across all plots were bimodal with modes at 0-4.9 cm and 25-29.9 cm in diameter. The highest mode was at 0-4.9 cm in treatment plots and at 25-29.9 cm in reference plots. Only fire quantitatively changed the distributions by the attrition of the smallest trees >1.4 m high in 1995 and 1997. Fire caused approximately 50% decreases in longleaf pine juvenile (<1.4 m high) density in 1995 and 1997. Longleaf pine juveniles were unaffected by other treatments. Juvenile densities were slightly higher and substantially more variable in reference plots than in treatments. Treatments had no effect on recently germinated longleaf pine seedlings. Densities ranged between 75 (control) and 105 seedlings/20 m^2 (felling/girdling) in treatment plots (15,182-21,255 seedlings/acre). Median reference plot densities were 205 seedlings/20 m^2 (41,498 seedling/acre). In 1997, fuel reduction burns decreased densities to <5 seedlings/20 m^2 (<1,012 seedlings/acre) in ULW® and felling/girdling plots. Seedling densities dropped by approximately 50% in control and burn plots, although these sites received no manipulations after 1995. Seedling densities only decreased by 22% in reference plots, which did experience some fires.

INTRODUCTION

Longleaf pine-dominated sandhills were once a dominant community type across much of the southeastern Coastal Plain (Frost 1993, Landers et al. 1995, Plunkett and Hall 1995). They were characterized by an open overstory of longleaf pine (*Pinus palustris*), with scattered hardwoods, such as turkey oak (*Quercus laevis*), and a diverse herbaceous understory (Myers 1990). Integral to the maintenance of community composition and

structure are fires, thought to have occurred historically every 1-10 years. Lightning and early human activities established this fire regime, which was seasonally biased toward the growing season months of April-July (Robbins and Myers 1992). Fires maintained the open structure of the community, top-killed many of the hardwoods which subsequently resprouted, stimulated flowering and seed production of several of the herbaceous species, created an ideal seedbed for recruitment of longleaf pine seedlings and herbaceous species, and constrained the distributions of fire-intolerant species.

During this century, prolonged fire suppression has resulted in hardwood encroachment to levels that threatened the recruitment of longleaf pine and herbaceous plants, and the viability of many federal- and state-listed endangered species. Returning fire to fire-suppressed sandhills is necessary to reverse degradation. However, the accumulation of woody fuels and fuel ladders represent a fire hazard to both the longleaf pine canopy and juveniles (Robbins and Myers 1992), and the lowered flammability of fuel requires more dangerous burn parameters for ignition success. Moreover, fire is not expected to topkill larger oaks (Waldrop et al. 1992, Streng et al. 1993), and topkilled oaks generally resprout in densities that would far exceed pre-burn levels in the absence of frequent fires (Waldrop et al. 1992). Therefore, techniques in addition to fire may be needed to kill larger hardwoods while preserving the longleaf pine component.

In this study, we experimentally compared the initial effects of three hardwood reduction techniques (growing season burn, application of the ULW® form of the herbicide hexazinone, and midstory mechanical felling/girdling) and a no-treatment control in fire-suppressed, longleaf pine-dominated sandhills on the densities of different diameter classes of longleaf pine and oaks. Sandhill communities that received treatments were also contrasted to frequently-burned longleaf pine-dominated sandhills, which were not part of the experimental design but are considered here as reference sites. Results presented here are from the first 4 years of the study: the 1994 pre-treatment phase and the 1995 and 1997 fall/winter seasons following treatment application. Although treatments were applied in 1995, ULW® and felling/girdling plots also were burned for fuel reduction during the winter 1996/1997.

We expected growing season fire to moderately reduce hardwood tree densities (Glitzenstein et al. 1995). We also expected approximately 50% mortality among smaller longleaf pines (2-5 cm DBH class) (Boyer 1990, Streng et al. 1993) because of their thinner bark and exposed terminal shoots. We predicted that the herbicide treatment would decrease hardwood tree and resprout densities (Brockway et al. 1998) more than the burn treatment. Chainsaw felling would greatly reduce hardwood densities, but promote oak resprouting. Mortality among smaller diameter classes of longleaf pines following fuel reduction burns in ULW® and felling/girdling plots should approximate levels seen in 1995 for growing season fire. A large fraction (>90%) of longleaf pine seedlings from the 1996 mast year were expected to be killed by fuel reduction burns (Grace and Platt 1995), although only 50% mortality of longleaf pine juveniles that germinated prior to 1996 was expected (Provencher et al. 1998).

METHODS

Study Area

Eglin Air Force Base (EAFB) occupies the southern portions of Walton, Okaloosa, and Santa Rosa Counties in the western Florida Panhandle (Fig. 3.1). The climate is temperate with mild winters and hot, humid summers. A more detailed description of weather, soils, and general botany is presented in Provencher et al. (1998) and in Rodgers and Provencher (1999).

Experimental Design

Selection of restoration plots and reference plots is described in Provencher et al. (1998). The experimental design is briefly presented. Reference plots are independent of the experimental design.

Restoration Blocks. We compared the initial effects of three hardwood reduction techniques and control (fire-suppression) on longleaf pine and oak densities in fire-suppressed, longleaf pine-dominated sandhills. (Responses of soil chemistry variables, other plants, and animals are presented in Chapter 4.) Treatments were growing season burning, application of the ULW® form of the herbicide hexazinone (75% active ingredient applied at a rate of 2.44 kg/ha [2 lb/acre]), and chainsaw felling/girdling. One replicate of each treatment (including the control) was randomly assigned to an 81-ha plot in one of six blocks in a randomized complete block design (Figs. 3.1 and 3.2) (Steel and Torrie 1980). Following pre-treatment sampling from spring 1994 to spring 1995, plots were burned in April, May, and June 1995. ULW® was applied by leaf-blower in early May, but the herbicide only became active in mid-May after abundant rains. Felling/girdling operations occurred between June and late August in the 20-ha (50-acre) sampling area of each plot. The majority of the remaining 61-ha (150-acre) portion of the plots was completed by August, but felling/girdling operations persisted until November in two plots. ULW® and felling/girdling plots were burned for fuel reduction from early March to late April 1997.

In each 81-ha plot, all subplots and sampling stations were located in the 20-ha corner farthest from the neighboring plots of the block in order to minimize inter-treatment effects (Fig. 3.2). We borrowed from split-plot terminology to label our nested sampling units: each plot contained 32, 10 × 40-m subplots (Fig. 3.2); any sampling unit within a subplot was referred to as a sub-subplot.

Reference Blocks. A total of six 81-ha frequently-burned, longleaf pine-dominated sandhill plots were established to represent objective goals for the restoration of fire-suppressed plots (Fig. 3.1: A-77, A-78, and B-75). Reference plots, although not part of the restoration experimental design, provide a benchmark for measurement of the success and efficacy of the restoration treatments applied. Each reference plot contained the same subplot sampling design as the restoration plots (Fig. 3.2), but the 20-ha sampling site was located in the plot centers to avoid potential edge effects.

Data Collection and Sampling

Pre-treatment tree densities, DBH, and height were collected from 1 November 1994 until 1 April 1995. Groundcover oak seedling densities were counted as part of the

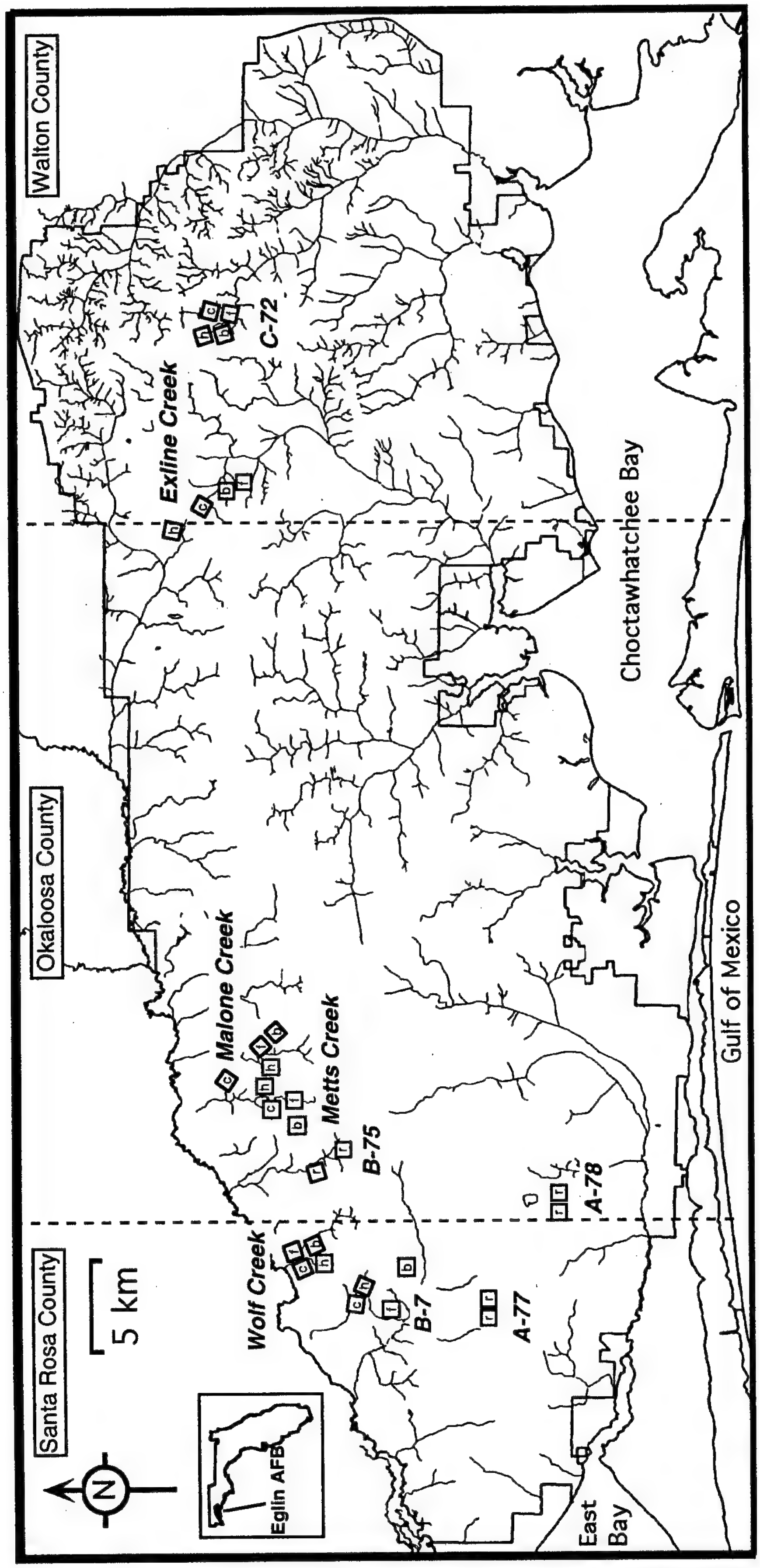

Fig. 3.1. Location of restoration and reference plots on Eglin Air Force Base, Florida. Small squares represent 81-ha (200-acre) plots. Treatment blocks include (from west to east) B-7, Wolf Creek, Metts Creek, Malone Creek, Exline Creek, and C-72. Reference blocks include (from west to east) A-77, A-78, and B-75. Legend: b = burn; c = control; f = felling/girdling; h = herbicide; r = reference.

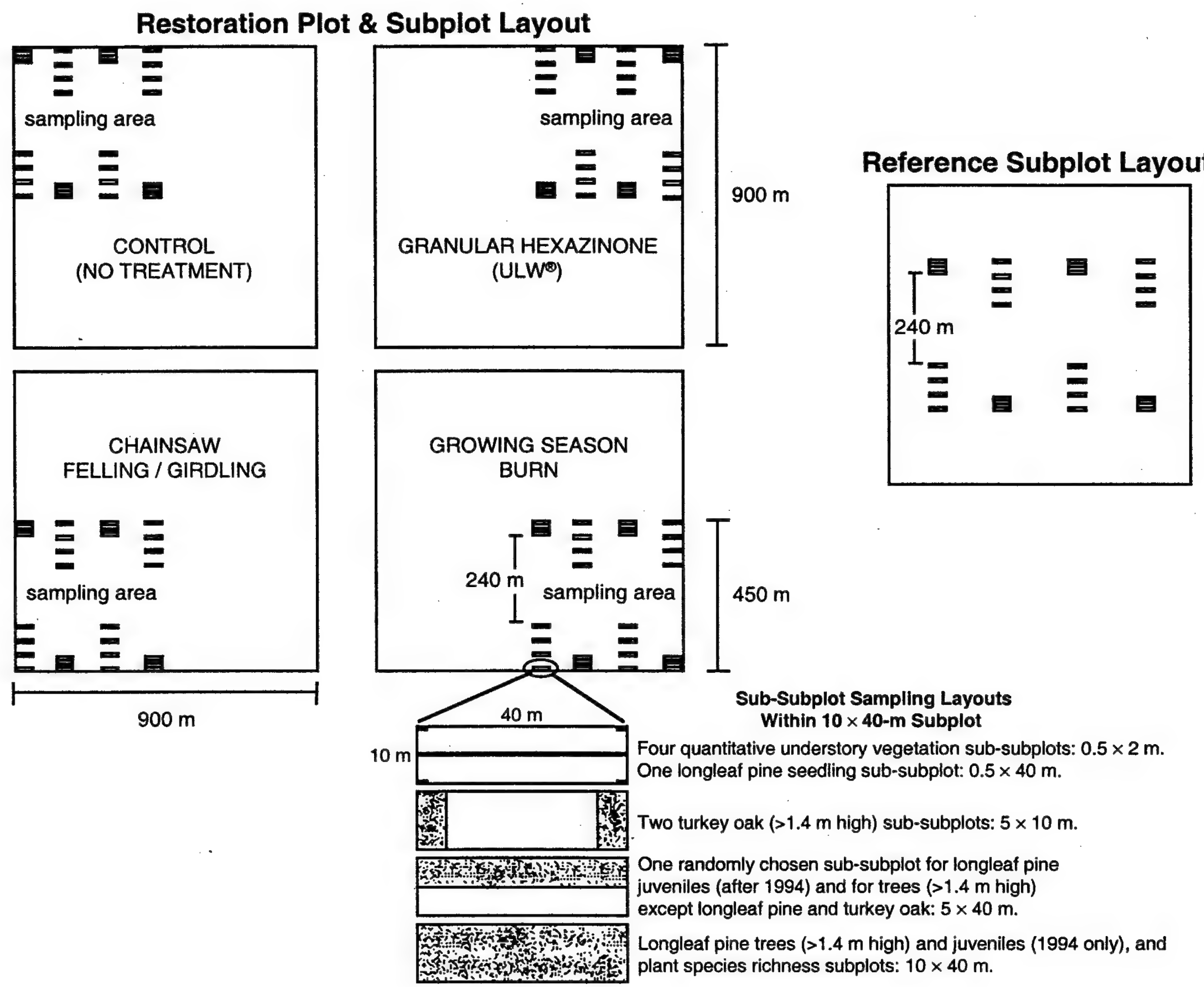

Fig. 3.2. Sample layout of 81-ha (200-acre) in one of six reference plots and in restoration plots and sampling areas in one of six blocks in a randomized complete block split-plot design consisting of four whole-plot treatments. Treatments were applied to the 81-ha plot. Spatial randomization of treatments varies per block. Subplot centers are separated by 10 or 50 m. One subplot and its four layers of sub-subplots are presented.

groundcover vegetation sampling from 8 July to 30 October 1994. The first post-treatment tree sampling period spanned from December 1995 until March 1996. The first late summer/fall post-treatment sampling of oak seedlings began immediately after treatment application and was completed by October 1995. Although longleaf pine juveniles and seedlings and oak seedlings were counted in the second year post-treatment, trees <1.4 m high were not sampled for DBH or height during fall 1996 and winter 1997. Third year post-treatment sampling closely matched the first year's schedule.

Based on preliminary analyses, we sub-sampled each 10 × 40-m subplot to facilitate collection of tree density, height, and DBH data. The area sampled was determined by evaluating variance components for dominant species in successively smaller units. Height and DBH of all longleaf pine within each 10 × 40-m subplot were measured (Fig. 3.2). Longleaf pine juveniles (<1.4 m high) were counted within a pre-determined, random longitudinal half of the 10 × 40-m area. (Hereafter, we use the term *juvenile* to describe longleaf pine individuals that range from the grass stage to <1.4 m height and that established prior to the fall 1996 mast year. *Seedlings* are those individuals that originated from the 1996 mast crop and that were recognizable by their blue-green color and a limited number of needles.) Longleaf pine seedlings were counted in a central rectangle measuring 0.5 × 40 m. Turkey oaks were sampled within two 5 × 10-m areas situated at the narrow ends of each 10 × 40-m subplot (Fig. 3.2). All other tree species were sampled in a randomly selected longitudinal half (i.e., 5 × 40-m) of each 10 × 40-m subplot (Fig. 3.2). Individual stems of groundcover oak seedlings (including tree species <1.4 m high) were counted in the four 0.5 × 2-m corner sub-subplots during the summer/fall vegetation sampling (Fig. 3.2).

We purged reference subplots that were not representative of the sandhill restoration subplots. We used canonical correspondence analysis (Kenkel and Orlóci 1986, Ter Braak 1986) and non-metric multidimensional scaling (Kruskall 1964, Kenkel and Orlóci 1986) on soil texture variables (percent total silt and clay and percent sand of different grain sizes) to identify these subplots as outliers (Provencher et al. 1998). Unfortunately, because percent total silt and clay was not the best predictor of subplot ordination, we could not remove subplots that were clearly akin to flatwoods due to underground water flow and very high silt and clay content. We found that the simple ranking of subplots by percent total silt and clay clearly identified mesic subplots often associated with underground sources of water or proximity to creeks and depressions. All reference subplots with >8% total silt and clay were discarded. Block A-78 west and east were the only plots that were affected.

Statistical Analyses

We graphed the medians, 25 and 75% quartiles, and minimum and maximum whole-plot averages of variables. (Three of six replicates closest to the median are contained within the 25 and 75% quartiles.) We chose to graph the median and 25 and 75% quartiles because they show the distribution of the data; however, the statistical tests described below and reported on the figures are based on means and variances.

We tested restoration treatment effects using a randomized complete block analysis of (co)variance (two-way ANOVA, ANCOVA) (Steel and Torrie 1980) for selected tree species variables in restoration plots. ANOVA was used for longleaf pine seedlings, because pre-treatment data did not exist (seedlings originated from the 1996 mast crop). Because of the availability of pre-treatment data for tree size distributions, longleaf pine juvenile density,

and oak juvenile density, we tested the effect of pre-treatment data on post-treatment data using ANCOVA. Pre-treatment data were used to adjust post-treatment averages to account for differences among treatments that existed prior to treatment application. The adjusted plot averages were the values used in the figures, except in tree size distribution figures, because of a management need to observe actual pre- and post-treatment values. (Thus, figures show the median of plot averages.) Multivariate analysis of covariance (MANCOVA) was used to test restoration effects on longleaf pine and oak size density distributions (only the four smallest 5-cm diameter classes had sufficient densities to allow testing), because shading and root competition may cause a correlation among diameter classes. This correlation violates the assumption of independent observations when several diameter classes are tested as independent variables. Despite this caveat, we tested treatment effects of all diameter classes by ANCOVA in order to generate contrasts per diameter class.

We calculated AN(C)OVAs using a computer randomization test (Edgington 1987). The details of the randomization procedure are described in Provencher et al. (1998). MANCOVA was executed with the software STATISTICA (1994).

Because it is only possible to perform a maximum number of independent contrasts that is equal to the degrees of freedom for restoration treatments (3 degrees of freedom) (Sokal and Rohlf 1981), we performed three independent contrasts to compare treatment means: control versus growing season burn (C vs. B), burn versus ULW® (B vs. U), and ULW® versus felling/girdling (U vs. F). The first contrast tested whether maintaining fire suppression (control) differed from burning. Burning is the management default at EAFB, because it is the least expensive management tool and because frequent fires characterize the original, more easily maintained condition of sandhills. Because felling/girdling and ULW® are the more expensive management techniques, their effects were compared to burning. These more expensive methods were also expected to reduce hardwood abundance more effectively than burning. We compared burning to ULW®, because these techniques differ greatly in the amount of oak resprouting and topkill mortality they cause. ULW® should practically eliminate oaks, whereas we expect only 50% topkill and abundant resprouting with fire (Glitzenstein et al. 1995). ULW® was compared to felling/girdling because of a management need to discriminate between equally expensive methods that differ in oak resprouting: felling oaks should trigger resprouting.

Most variables needed transformation because they displayed non-normal distributions and heterogeneous variances. Logarithmic transformations (log[X + 1]) were applied to all densities because they showed significant and positive mean-variance relationships (Sokal and Rohlf 1981). For simplicity and ease of reading, we have termed the tests of restoration treatments in the statistical tables as "restoration". We refer to the multivariate effect of restoration treatments as "multivariate restoration effect" in the text.

RESULTS

Oak Size Distributions

Pre-treatment oak size distributions followed a negative-exponential curve in all restoration treatments (Fig. 3.3). The size distribution in reference plots was practically flat with low tree densities in all diameter classes, although densities somewhat decreased with increasing size (Fig. 3.3).

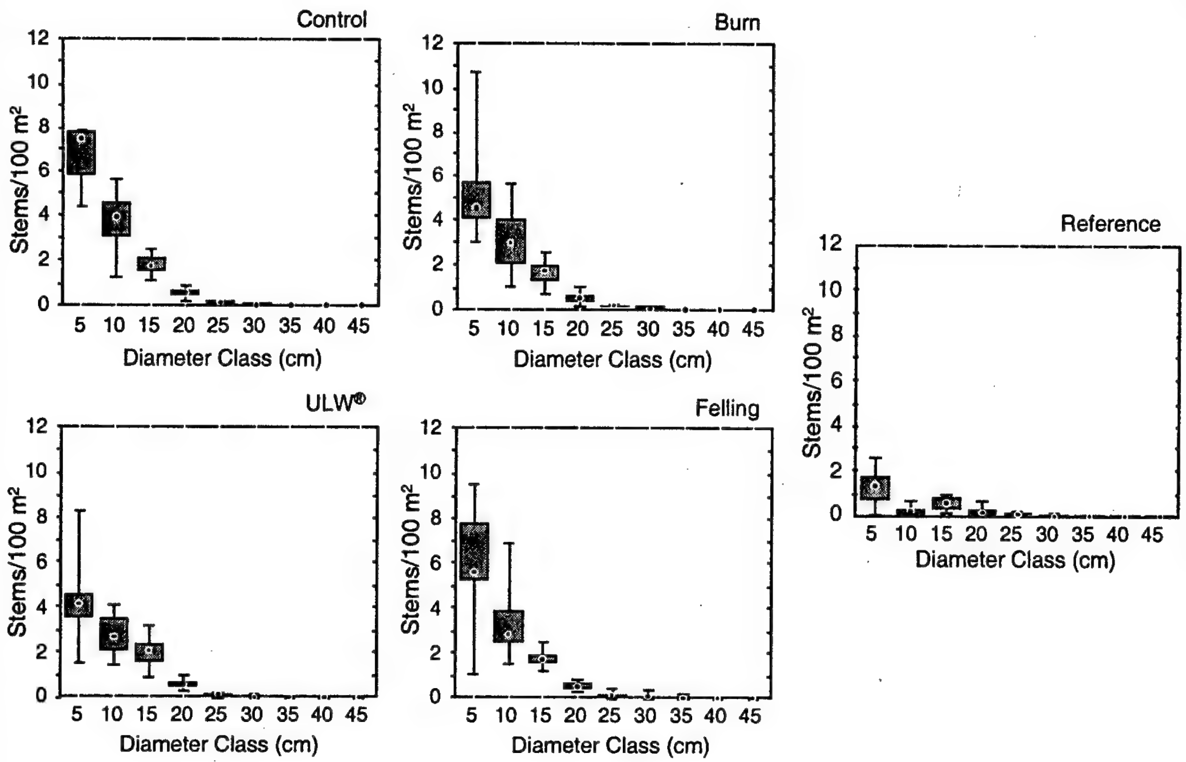

Fig. 3.3. Pre-treatment (1994) density of all oak species per 5 cm diameter classes in restoration and reference plots. Other hardwoods not presented. Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

In 1995, multivariate restoration effects and pre-treatment effects on the four smallest diameter classes were highly significant (Table 3.1). Univariate tests show that densities in the five smallest diameter classes were significantly reduced by >60% in ULW® and by >90% in felling/girdling plots, although these percentages varied with diameter classes (Fig. 3.4, Table 3.1). The negative effect of felling/girdling on densities was significantly greater than that of ULW® application only for the smallest oaks (Table 3.1). Burning resulted in significantly more topkill than the control treatment for trees in the 0-4.9 cm and 15-19.9 cm classes. In both cases, topkill was less than 20% based on medians compared to pre-treatment densities (Figs. 3.5 vs. 3.4). Burning topkilled significantly fewer oaks than ULW® treatments (Table 3.1).

The multivariate restoration effect persisted in 1997, but the pre-treatment effect was marginally significant ($P = 0.0524$, Table 3.1). Oak densities were approximately 50% smaller for the 0-4.9, 5-9.9 cm, and 10-14.9 cm DBH classes in 1997 compared to 1994, which indicates a delayed mortality response by oaks to burning (Fig. 3.5). Densities were significantly lower in these diameter classes in burn plots compared to control plots, but not in the 20-24.9 cm class (Table 3.1). There was no significant univariate difference between ULW® and felling/girdling plots after fuel reduction burns (Table 3.1). These burns, however, may have reduced resprout densities in the 0-4.9 cm class of the felling/girdling plots (Fig. 3.5). Large numbers of resprouts were observed in the field prior to fuel reduction burns.

Table 3.1. Two-way MANCOVA and ANCOVA for tests of restoration treatments and pre-treatment effects on densities of oaks by 5 cm diameter classes from 1995 to 1997 in mixed hardwood and longleaf pine forests at Eglin Air Force Base, Florida. Restoration treatments are growing season burn, application of ULW® herbicide, hand felling and girdling of hardwoods and sand pine, and no-treatment control. The experimental design is a randomized complete block, split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994. The error term is the mean square of the interaction of the block and restoration effects. MANCOVA was performed with STATISTICA (1994). For univariate tests, significance probabilities and sum of squares were calculated by a computer randomization ANCOVA based on 10,000 permutations. Calculations and tests followed Steel and Torrie (1980: 215-217, 260, 411-419). Tree sizes were log(X+1)-transformed to stabilize variances.

Multivariate Tests

Year	Source	*Wilks' λ*	df	*P*
1995	Pre-treatment	0.0218	16, 25	0.0011
	Restoration	0.0027	12, 21	<0.00001
1997	Pre-treatment	0.0778	16, 25	0.0524
	Restoration	0.0282	12, 21	0.0006

Univariate Tests

Diameter class	Source	Sum of squares	*t*	df	*P*
1995					
0-4.9 cm	Block	0.3171		5	
	Restoration	14.6234		3	0.0000
	Pre-treatment	1.1142		1	0.0025
	Error	1.1246		14	
	Contrast				
	C vs. B†		0.8394	1	0.0204
	B vs. U		1.7274	1	0.0008
	U vs. F		1.4020	1	0.0005
5-9.9 cm	Block	1.6490		5	
	Restoration	7.8306		3	0.0000
	Pre-treatment	0.7667		1	0.0025
	Error	0.6873		14	
	Contrast				
	C vs. B		0.3737	1	0.1582
	B vs. U		2.4411	1	0.0000
	U vs. F		0.3569	1	0.0833
10-14.9 cm	Block	0.5527		5	
	Restoration	3.6349		3	0.0000
	Pre-treatment	0.3345		1	0.0025
	Error	0.2786		14	
	Contrast				
	C vs. B		0.4355	1	0.1265
	B vs. U		3.2120	1	0.0001
	U vs. F		−0.0183	1	0.4731
15-19.9 cm	Block	0.1738		5	
	Restoration	0.3071		3	0.0000
	Pre-treatment	0.0227		1	0.0250

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.

Table 3.1. Continued.

Diameter class	Source	Sum of squares	t	df	P
	Error	0.0400		14	
	Contrast				
	C vs. B		1.0996	1	0.0077
	B vs. U		2.2444	1	0.0000
	U vs. F		− 0.1017	1	0.3399
20-24.9 cm	Block	0.0790		5	
	Restoration	0.0114		3	0.1419
	Pre-treatment	0.0453		1	0.0025
	Error	0.0271		14	
1997					
0-4.9 cm	Block	0.7702		5	
	Restoration	9.8756		3	0.0025
	Pre-treatment	0.3230		1	0.5000
	Error	5.0247		14	
	Contrast				
	C vs. B		0.9158	1	0.0152
	B vs. U		0.9548	1	0.0119
	U vs. F		− 0.3863	1	0.1288
5-9.9 cm	Block	1.6755		5	
	Restoration	6.2071		3	0.0000
	Pre-treatment	0.9156		1	0.0025
	Error	0.8906		14	
	Contrast				
	C vs. B		1.4192	1	0.0018
	B vs. U		1.5130	1	0.0014
	U vs. F		0.1358	1	0.3034
10-14.9 cm	Block	0.7858		5	
	Restoration	4.3451		3	0.0000
	Pre-treatment	0.4260		1	0.0025
	Error	0.4616		14	
	Contrast				
	C vs. B		1.2422	1	0.0028
	B vs. U		2.2671	1	0.0001
	U vs. F		− 0.0838	1	0.3832
15-19.9 cm	Block	0.2311		5	
	Restoration	0.5016		3	0.0000
	Pre-treatment	0.0041		1	0.5000
	Error	0.1066		14	
	Contrast				
	C vs. B		1.1087	1	0.0059
	B vs. U		1.5694	1	0.0006
	U vs. F		− 0.1169	1	0.3222
20-24.9 cm	Block	0.0404		5	
	Restoration	0.0171		3	0.0207
	Pre-treatment	0.0370		1	0.0025
	Error	0.0185		14	
	Contrast				
	C vs. B		0.0542	1	0.4352
	B vs. U		1.1274	1	0.0021
	U vs. F		− 0.1231	1	0.2398

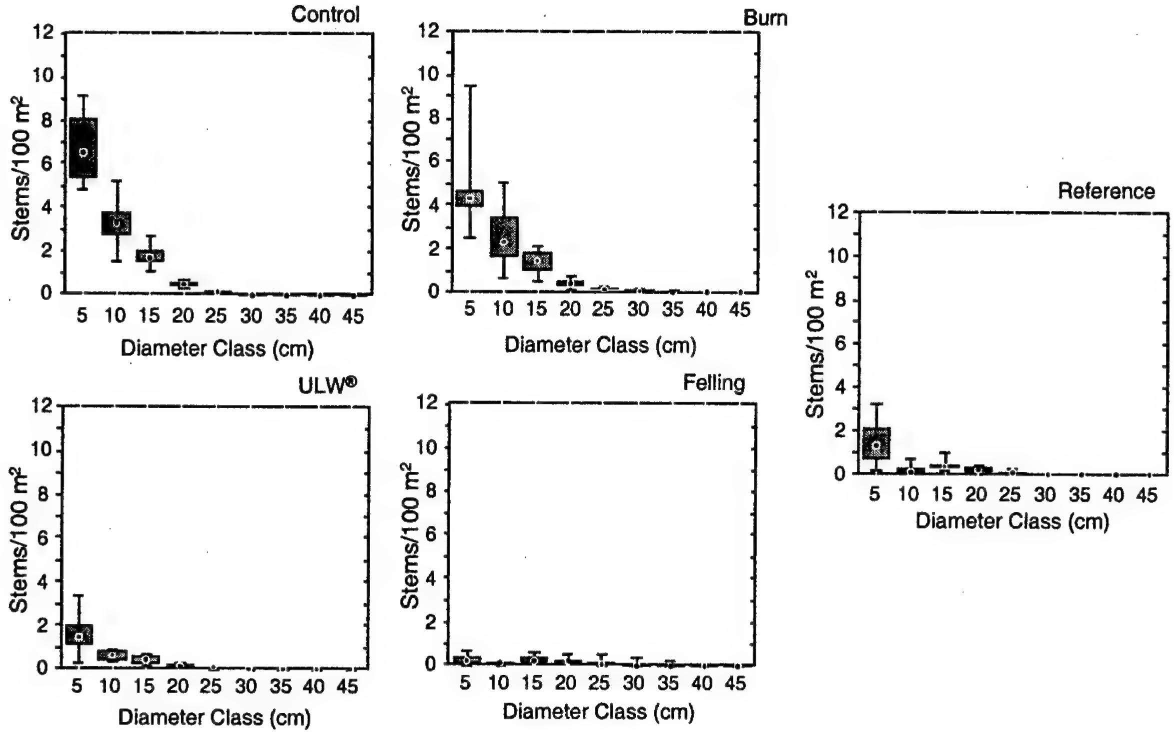

Fig. 3.4. First year post-treatment (1995) density of oaks per 5 cm diameter classes in restoration and reference plots. Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum values. Sample size = 6.

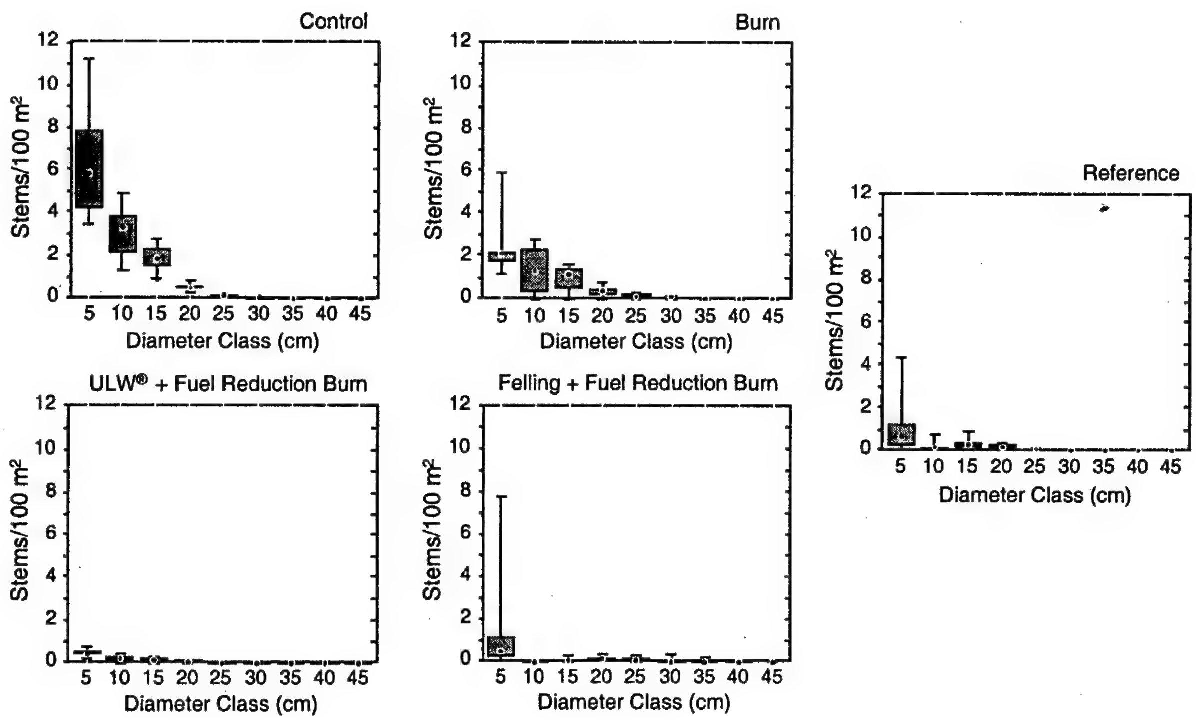

Fig. 3.5. Third year post-treatment (1997) density of oaks per 5 cm diameter classes in restoration and reference plots. Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum values. Sample size = 6.

Longleaf Pine Size Distributions

Size distributions of longleaf pine were similar among restoration treatments in the pre-treatment phase. Distributions were bimodal with peaks at 0-4.9 cm and at 25-29.9 cm (Fig. 3.6). The highest mode was at 0-4.9 cm in all treatment plots, but at 25-29.9 cm in reference plots (Fig. 3.6).

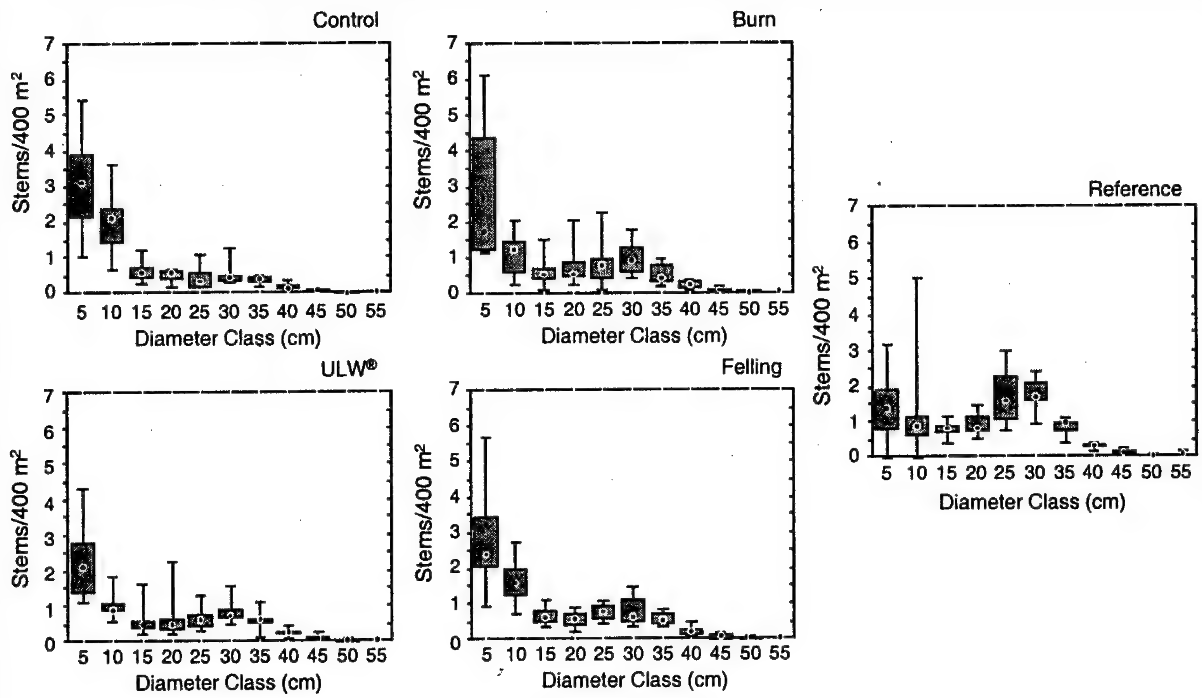

Fig. 3.6. Pre-treatment (1994) density of longleaf pines per 5 cm diameter classes in restoration and reference plots. Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

The qualitative shapes of distributions and position of modes remained the same for all years post-treatment (Figs. 3.7-3.8). Highly significant pre-treatment effects from MANCOVA support this observation (Table 3.2). Only fire quantitatively changed the distributions by the attrition of the smallest trees; however, this difference was not significant in the MANCOVA (Table 3.2). The only significant univariate test for trees 0-4.9 cm confirmed the negative effect of fire (Table 3.2).

In 1997, dormant season fuel reduction burns in ULW® and felling/girdling plots once again decreased longleaf pine densities in the two smallest diameter classes (Fig. 3.8). The MANCOVA restoration effect was significant in 1997, and significant univariate restoration effects extended from the 0-4.9 cm to the 5-9.9 cm class (Table 3.2). Although fire following ULW® application was significantly less harmful to trees than growing season fire applied in 1995, we could not directly test the same relationship for felling/girdling plots (Table 3.2). The *t*-values from the contrasts suggest that felling/girdling is more harmful to these diameter classes than ULW® application when both treatments were followed by fuel-reduction fire.

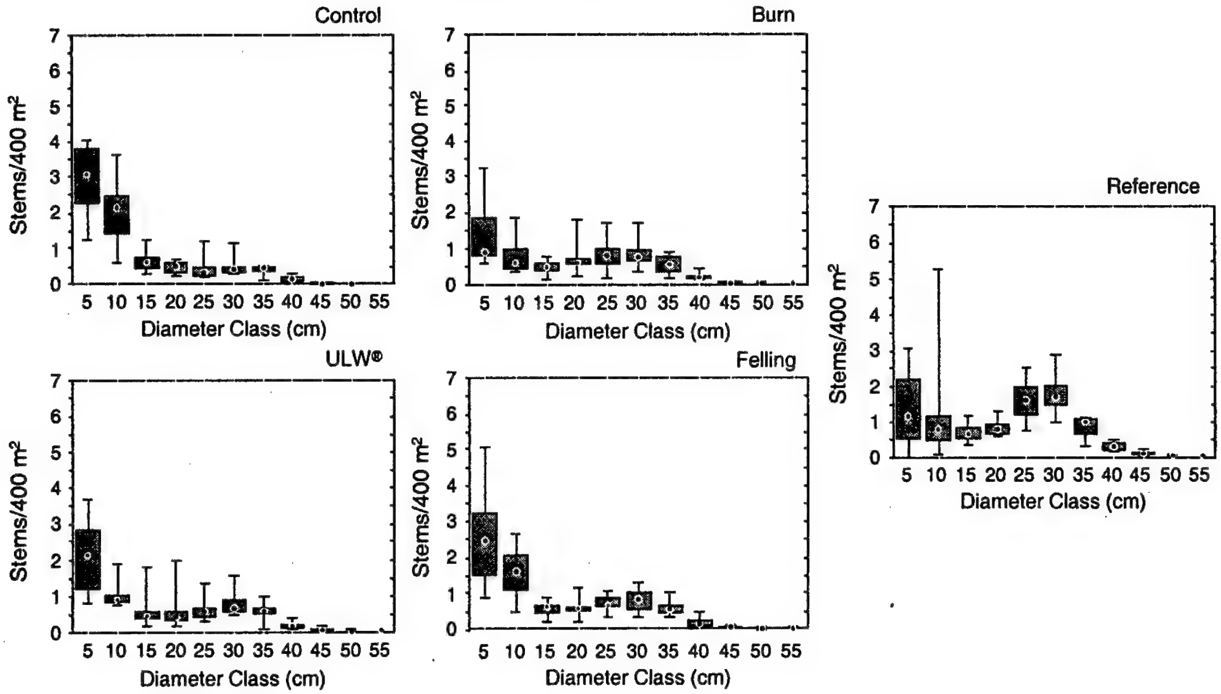

Fig. 3.7. First year post-treatment (1995) density of longleaf pines per 5 cm diameter classes in restoration and reference plots. Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

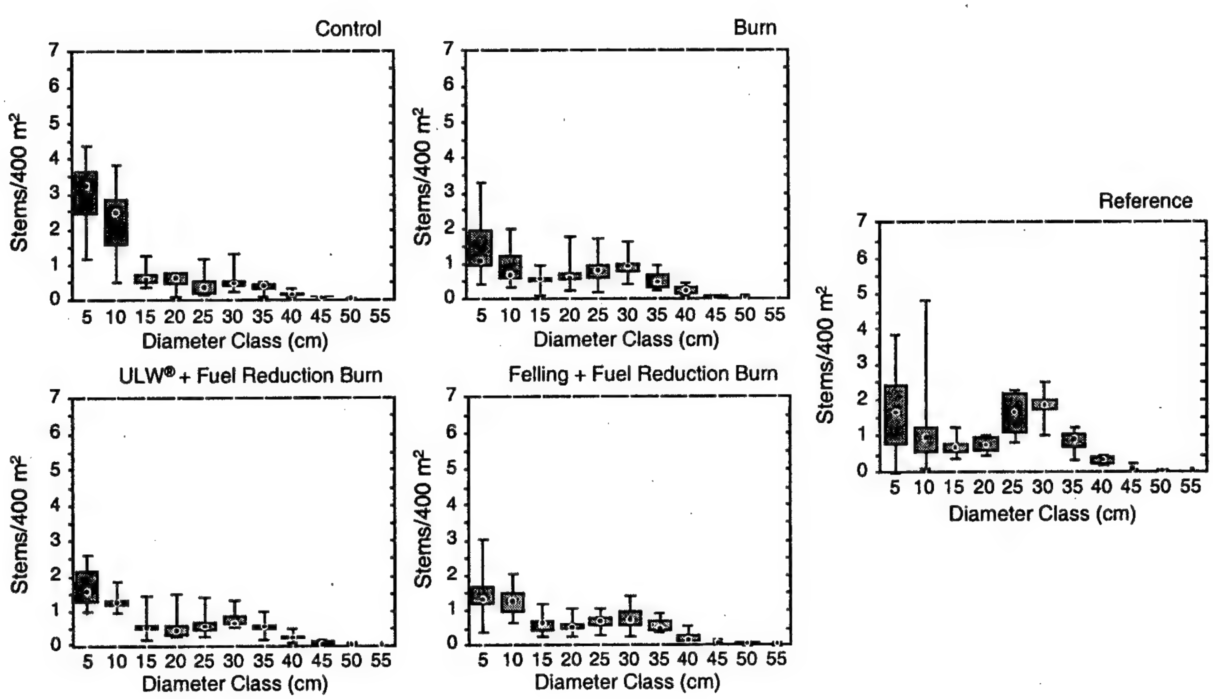

Fig. 3.8. Third year post-treatment (1997) density of longleaf pines per 5 cm diameter classes in restoration and reference plots. Fuel reduction burns were applied in ULW® and felling plots. Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

Table 3.2. Two-way MANCOVA and ANCOVA for tests of restoration treatments and pre-treatment effects on densities of longleaf pine by 5 cm diameter classes from 1995 to 1997 in mixed hardwood and longleaf pine forests at Eglin Air Force Base, Florida. Restoration treatments are growing season burn, application of ULW® herbicide, hand felling and girdling of hardwoods and sand pine, and no-treatment control. The experimental design is a randomized complete block, split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994. The error term is the mean square of the interaction of the block and restoration effects. MANCOVA was performed with STATISTICA (1994). For univariate tests, significance probabilities and sum of squares were calculated by a computer randomization ANCOVA based on 10,000 permutations. Calculations and tests followed Steel and Torrie (1980: 215-217, 260, 411-419). Tree sizes were log(X+1)-transformed to stabilize variances.

Multivariate Tests

Year	Source	*Wilks' λ*	df	*P*
1995	Pre-treatment	0.0001	16, 25	<0.00001
	Restoration	0.1540	12, 21	0.1071
1997	Pre-treatment	0.0012	16, 25	<0.00001
	Restoration	0.0909	12, 21	0.0249

Univariate Tests

Diameter class	Source	Sum of squares	*t*	df	*P*
1995					
0-4.9 cm	Block	0.4037		5	
	Restoration	0.2160		3	0.0219
	Pre-treatment	0.4754		1	0.0025
	Error	0.2335		14	
	Contrast				
	C vs. B†		1.0910	1	0.0048
	B vs. U		– 1.1319	1	0.0040
	U vs. F		0.0888	1	0.3514
5-9.9 cm	Block	0.1813		5	
	Restoration	0.0279		3	0.1970
	Pre-treatment	0.1978		1	0.0025
	Error	0.0773		14	
10-14.9 cm	Block	0.0670		5	
	Restoration	0.0072		3	0.2228
	Pre-treatment	0.0482		1	0.0025
	Error	0.0206		14	
15-19.9 cm	Block	0.1041		5	
	Restoration	0.0010		3	0.4965
	Pre-treatment	0.0882		1	0.0025
	Error	0.0054		14	
20-24.9 cm	Block	0.1049		5	
	Restoration	0.0009		3	0.4814
	Pre-treatment	0.0711		1	0.0025
	Error	0.0050		14	

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.

Table 3.2. Continued.

Diameter class	Source	Sum of squares	*t*	df	*P*
25-29.9 cm	Block	0.0714		5	
	Restoration	0.0017		3	0.7462
	Pre-treatment	0.0725		1	0.0025
	Error	0.0158		14	
30-34.9 cm	Block	0.0450		5	
	Restoration	0.0002		3	0.8250
	Pre-treatment	0.0184		1	0.0025
	Error	0.0023		14	
35-39.9 cm	Block	0.0059		5	
	Restoration	0.0000		3	0.9640
	Pre-treatment	0.0086		1	0.0025
	Error	0.0015		14	
40-44.9 cm	Block	0.0007		5	
	Restoration	0.0000		3	0.8178
	Pre-treatment	0.0008		1	0.0025
	Error	0.0003		14	
1997					
0-4.9 cm	Block	0.1762		5	
	Restoration	0.2529		3	0.0010
	Pre-treatment	0.3864		1	0.0025
	Error	0.1372		14	
	Contrast				
	C vs. B		1.4539	1	0.0005
	B vs. U		– 0.8804	1	0.0166
	U vs. F		0.3105	1	0.1032
5-9.9 cm	Block	0.1580		5	
	Restoration	0.0760		3	0.0211
	Pre-treatment	0.1572		1	0.0025
	Error	0.0855		14	
	Contrast				
	C vs. B		0.5195	1	0.0833
	B vs. U		– 0.8692	1	0.0117
	U vs. F		0.2234	1	0.1465
10-14.9 cm	Block	0.0592		5	
	Restoration	0.0047		3	0.4460
	Pre-treatment	0.0314		1	0.0025
	Error	0.0236		14	
15-19.9 cm	Block	0.0829		5	
	Restoration	0.0011		3	0.7317
	Pre-treatment	0.0577		1	0.0025
	Error	0.0121		14	
20-24.9 cm	Block	0.1064		5	
	Restoration	0.0003		3	0.9612
	Pre-treatment	0.0625		1	0.0025
	Error	0.0119		14	
25-29.9 cm	Block	0.0821		5	
	Restoration	0.0002		3	0.9577
	Pre-treatment	0.0599		1	0.0025
	Error	0.0111		14	
30-34.9 cm	Block	0.0420		5	
	Restoration	0.0001		3	0.9055
	Pre-treatment	0.0215		1	0.0025

Table 3.2. Continued.

Diameter class	Source	Sum of squares	*t*	df	*P*
	Error	0.0026		14	
35-39.9 cm	Block	0.0105		5	
	Restoration	0.0002		3	0.8308
	Pre-treatment	0.0117		1	0.0025
	Error	0.0028		14	
40-44.9 cm	Block	0.0013		5	
	Restoration	0.0002		3	0.1804
	Pre-treatment	0.0013		1	0.0025
	Error	0.0004		14	

Oak Juveniles

In both 1995 and 1996, control plots contained significantly fewer oak juveniles than the burn plots, both had significantly more juveniles than the ULW® plots (Fig. 3.9, Table 3.3). Reference plots contained the lowest and most variable median juvenile densities. The general pattern of densities persisted in 1997, but with minor differences. Densities were not significantly different between control and burn plots in 1997 (Table 3.3). Also, felling/girdling plots contained significantly more juveniles than the ULW® plots, and perhaps, the other plots (Fig. 3.9, Table 3.3).

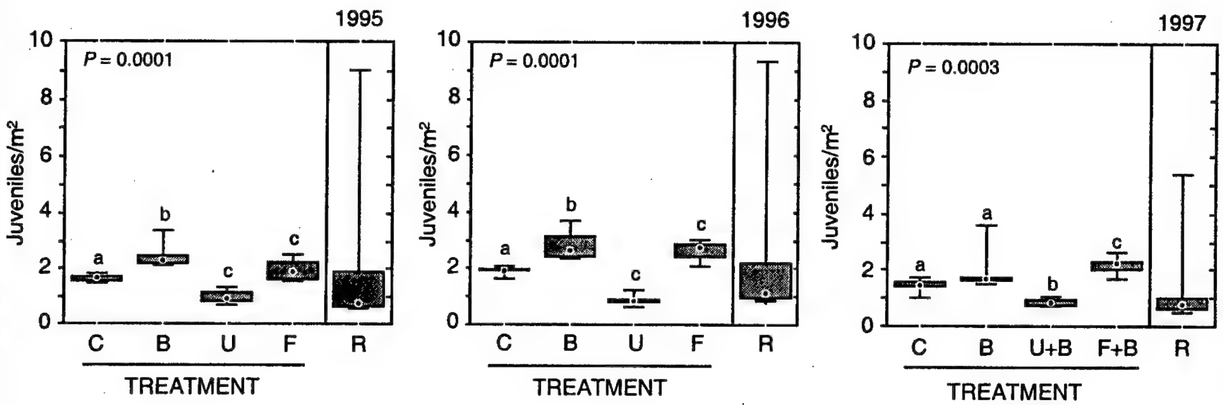

Fig. 3.9. Number of all oak species <1.4 m height/m^2 in restoration and reference plots post-treatment (1995-1997). Fuel reduction burns were applied in 1997. Oak densities were adjusted for restoration treatments only and were estimated from pre-treatment values using ANCOVA. Center of box represents the median, upper and lower edges of box are the 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Significance probability is the test of the effects of the four restoration treatments, which do not include the R plots. Lowercase letters associated with error bars code for the three following independent contrasts: C vs. B, B vs. U, and U vs. F. Different letters indicate significantly different means. Legend: B = burn; C = control; F = felling/girdling; R = reference; U = ULW® herbicide.

Table 3.3. Two-way analyses of (co)variance (AN[C]OVA) for tests of restoration treatments and pre-treatment effects on densities of oak juveniles (<1.4 m height) from 1995 to 1997 in mixed hardwood and longleaf pine forests at Eglin Air Force Base, Florida. Restoration treatments are growing season burn, application of ULW® herbicide, hand felling and girdling of hardwoods and sand pine, and no-treatment control. The experimental design is a randomized complete block, split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994. The error term is the mean square of the interaction of the block and restoration effects. Significance probabilities and sum of squares were calculated by a computer randomization ANCOVA based on 10,000 permutations. Calculations and tests followed Steel and Torrie (1980: 215-217, 260, 411-419). Tree densities were log(X+1)-transformed to stabilize variances. See also Fig. 3.9.

Year	Source	Sum of squares	t	df	P
1995	Block	0.4452		5	
	Restoration	0.9988		3	0.0000
	Pre-treatment	0.3063		1	0.0025
	Error	0.1872		14	
	Contrast				
	C vs. B†		– 1.2793	1	0.0020
	B vs. U		3.1942	1	0.0000
	U vs. F		– 0.3872	1	0.1325
1996	Block	0.4387		5	
	Restoration	1.8582		3	0.0000
	Pre-treatment	0.4901		1	0.0025
	Error	0.2196		14	
	Contrast				
	C vs. B		– 1.0835	1	0.0051
	B vs. U		3.8719	1	0.0000
	U vs. F		– 0.4705	1	0.0879
1997	Block	0.2590		5	
	Restoration	0.8845		3	0.0003
	Pre-treatment	0.6746		1	0.0025
	Error	0.2254		14	
	Contrast				
	C vs. B		– 0.5819	1	0.0607
	B vs. U		2.5409	1	0.0000
	U vs. F		– 0.5747	1	0.0510

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.

Longleaf Pine Juveniles

The adjusted densities of longleaf pine juveniles among treatments in all years post-treatment again reflect the critical role of fire. In 1995, growing season fire significantly decreased median density by approximately 50% to a value of 9 juveniles/200 m^2 (182.1 juveniles/acre) compared to other treatments (Fig. 3.10, Table 3.4). Longleaf pine juveniles were unaffected by other treatments. Median juvenile densities were slightly higher and substantially more variable in reference plots than in treatments (Fig. 3.10). The pattern of 1995 remained unchanged into 1996 (Fig. 3.10, Table 3.4), with the exception that median density in reference plots was comparable to that of non-burn plots (13-14 juveniles/200 m^2 or 263.1-283.4 juveniles/acre) and less variable than the previous year. Following fuel reduction burns in ULW® and felling/girdling plots in 1997, median adjusted density decreased to approximately 5-6 juveniles/200 m^2 (101.2-121.4 juveniles/acre) in all plots, including the control. Slight treatment differences were significant because of very low variability: control and ULW® plots contained significantly more juveniles than burn and felling/girdling plots (Table 3.4).

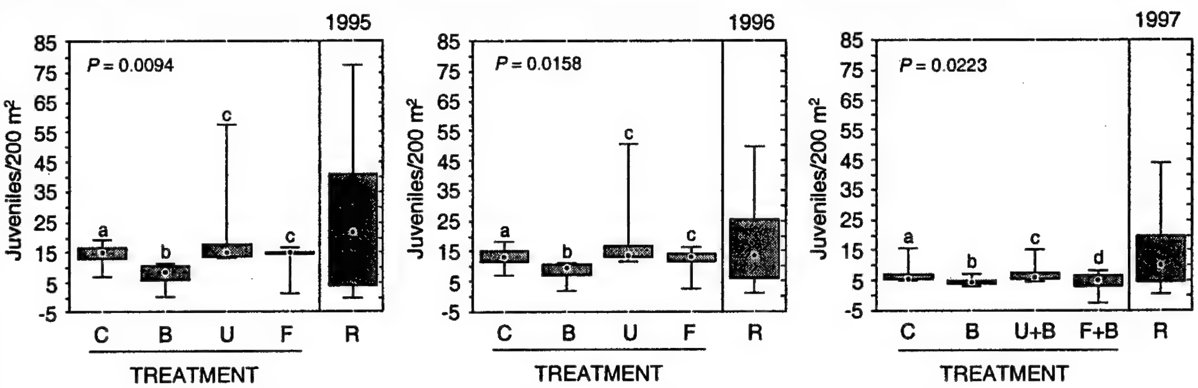

Fig. 3.10. Number of longleaf pine <1.4 m height/m^2 (except seedlings germinated in 1996) in restoration and reference plots post-treatment (1995-1997). Fuel reduction burns were applied in 1997. Pine densities were adjusted for restoration treatments only and were estimated from pre-treatment values using ANCOVA. Center of box represents the median, upper and lower edges of box are the 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Significance probability is the test of the effects of the four restoration treatments, which do not include the R plots. Lowercase letters associated with error bars code for the three following independent contrasts: C vs. B, B vs. U, and U vs. F. Different letters indicate significantly different means. Legend: B = burn; C = control; F = felling/girdling; R = reference; U = ULW® herbicide.

Table 3.4. Two-way analyses of (co)variance (AN[C]OVA) for tests of restoration treatments and pre-treatment effects on densities of longleaf pine juveniles (<1.4 m height) from 1995 to 1997 in mixed hardwood and longleaf pine forests at Eglin Air Force Base, Florida. Restoration treatments are growing season burn, application of ULW® herbicide, hand felling and girdling of hardwoods and sand pine, and no-treatment control. The experimental design is a randomized complete block, split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994. The error term is the mean square of the interaction of the block and restoration effects. Significance probabilities and sum of squares were calculated by a computer randomization ANCOVA based on 10,000 permutations. Calculations and tests followed Steel and Torrie (1980: 215-217, 260, 411-419). Tree densities were log(X+1)-transformed to stabilize variances. See also Fig. 3.10.

Year	Source	Sum of squares	*t*	df	*P*
1995	Block	15.4631		5	
	Restoration	7.8045		3	0.0094
	Pre-treatment	7.5909		1	0.0025
	Error	6.9683		14	
	Contrast				
	C vs. B†		1.0647	1	0.0033
	B vs. U		– 1.3227	1	0.0014
	U vs. F		0.3631	1	0.1316
1996	Block	14.4328		5	
	Restoration	5.6262		3	0.0158
	Pre-treatment	6.9266		1	0.0025
	Error	5.6813		14	
	Contrast				
	C vs. B		0.8936	1	0.0107
	B vs. U		– 1.2843	1	0.0016
	U vs. F		0.3823	1	0.1167
1997	Block	7.0865		5	
	Restoration	2.2566		3	0.0223
	Pre-treatment	4.5862		1	0.0025
	Error	2.6833		14	
	Contrast				
	C vs. B		0.9319	1	0.0086
	B vs. U		– 0.8317	1	0.0161
	U vs. F		0.6024	1	0.0314

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.

Longleaf Pine Seedlings

Median seedling densities did not differ among treatments in 1996 (Fig. 3.11, Table 3.5). Densities ranged between approximately 75 (control) and 105 seedlings/20 m^2 (felling/girdling) in treatment plots (15,182 and 21,255 seedlings/acre, respectively). Reference plot densities were approximately 205 seedlings/20 m^2 (21,255 seedlings/acre). Fuel reduction burns in 1997 decreased median densities by 95% to <5 seedlings/20 m^2 (1,012 seedlings/acre) in ULW® and felling/girdling plots. Seedling densities dropped by approximately 50% in control and burn plots, which were not manipulated after 1995.

Seedling densities only decreased by only 22% in reference plots, which did experience some fires.

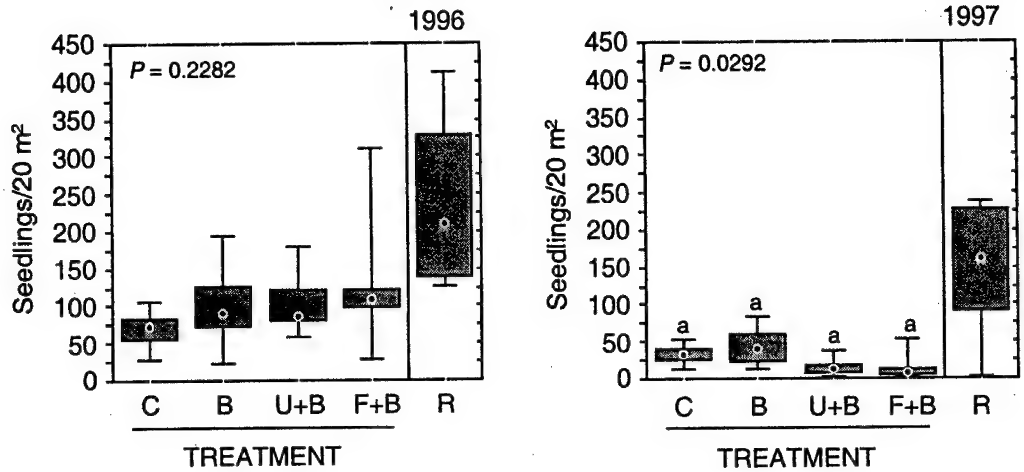

Fig. 3.11. Number of longleaf pine seedlings/m^2 (germinated in 1996) in restoration and reference plots post-treatment (1996-1997). Fuel reduction burns were applied in 1997. Seedling densities were adjusted for restoration treatments only and were estimated from pre-treatment values using ANCOVA. Center of box represents the median, upper and lower edges of box are the 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Significance probability is the test of the effects of the four restoration treatments, which do not include the R plots. Lowercase letters associated with error bars code for the three following independent contrasts: C vs. B, B vs. U, and U vs. F. Different letters indicate significantly different means. Legend: B = burn; C = control; F = felling/girdling; R = reference; U = ULW® herbicide.

Table 3.5. Two-way analyses of (co)variance (AN[C]OVA) for tests of restoration treatments and pre-treatment effects on densities of longleaf pine seedlings from 1996 to 1997 in mixed hardwood and longleaf pine forests at Eglin Air Force Base, Florida. Restoration treatments are growing season burn, application of ULW® herbicide, hand felling and girdling of hardwoods and sand pine, and no-treatment control. The experimental design is a randomized complete block, split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994. The error term is the mean square of the interaction of the block and restoration effects. Significance probabilities and sum of squares were calculated by a computer randomization ANCOVA based on 10,000 permutations. Calculations and tests followed Steel and Torrie (1980: 215-217, 260, 411-419). Tree densities were log(X+1)-transformed to stabilize variances. See also Fig. 3.11.

Year	Source	Sum of squares	*t*	df	*P*
1996	Block	10.5143		5	
	Restoration	2.5080		3	0.2282
	Error	7.4673		15	
1997	Block	5.8919		5	
	Restoration	18.6561		3	0.0292
	Error	20.8831		15	
	Contrast				
	C vs. B†		– 0.4981	1	0.4590
	B vs. U		3.3878	1	0.0930
	U vs. F		0.8642	1	0.4430

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.

DISCUSSION

Oak Size Distributions

We reported that hardwood size distributions in all plots closely followed the negative-exponential curve (Fig. 3.3). Qualitatively, Waldrop et al. (1992) also obtained this curve prior to initiating a long-term burn study at the Santee Fire Study Plots in the Francis Marion National Forest of South Carolina. We would expect this type of size distributions for shade-tolerant species such as hardwoods (Oliver and Larson 1990).

Hardwood reduction techniques varied in their effectiveness. Felling/girdling and ULW® application were both effective at topkilling most oaks. Neary et al. (1981), McLemore (1983), and Brockway et al. (1998) reported hardwood mortality values of 83-96%, 72-86%, and 83%, respectively, due to ULW®. These mortality values were dependent on soils, hardwood composition, and application rates. We found a mortality value of 78% for turkey oak stem density one year post-treatment in ULW® plots (Provencher et al. 1998). Herbicide-induced mortality was 55.5% for sand live oak (*Quercus geminata*) stem density (Provencher et al. 1998). In a recent study of hexazinone at EAFB, Berish (1996) found only 53% oak mortality caused by ULW® and 40% for brushbullet (Pronone®) one year after treatment. The higher density of sand live oak in Berish's (1996) plots compared to those reported here may explain his lower mortality results.

Growing season burning topkilled only 20% of all oaks one year post-treatment. Oak topkill mortality caused by fire is usually stronger for the smallest diameter classes (Waldrop et al. 1992, Rebertus et al. 1993), but does not generally exceed 58% for all diameter classes combined (Rebertus et al. 1989a,b, Glitzenstein et al. 1995). We believe that the topkill mortality reported here was low because fire intensity was low in one or more growing season burns.

Both burning and felling oaks necessitate follow-up growing season fires to control hardwood resprouting. Hardwood resprout densities can reach levels exceeding those found in the pre-burn state and control plots (Waldrop et al. 1992). We observed a wide range of increasing densities of resprouts (0-4.9 cm) among felling/girdling plots one year after these plots were burned in 1997 (Figs. 3.4 vs. 3.5). We expect even higher densities of resprouts for the 1998 sample.

We did not anticipate that the greatest effect of the growing season burning of 1995 on oak topkill mortality would manifest itself in 1997 (Fig. 3.4 vs. Fig. 3.5). Delayed mortality was approximately 50% for trees <15 cm DBH. Because most published experimental studies (Waldrop et al. 1992, Glitzenstein et al. 1995) of fire effects on hardwoods involve repeated burns, we could not find data on delayed hardwood mortality following a single burn. The severe summer drought in 1997 and several hurricanes may have killed already stressed trees belonging to the three smaller diameter classes.

Longleaf Pine Size Distributions

Bimodality in longleaf pine size distributions was reported at the Wade Tract of southern Georgia (Platt, Evans, and Rathbun 1988) and at the Escambia Experimental Forest of southern Alabama (Boyer 1990, Farrar 1993). Their highest mode was for the smallest diameter classes, which we also reported for fire-suppressed stands. Platt, Evans, and Rathbun (1988) and Farrar (1993) detected a second mode for trees 40-49.9 cm in diameter, whereas we detected a second mode at 25-29.9 cm, which was dominant only in reference plots. The Wade Tract and the Escambia Experimental Forest are frequently-burned forests. Therefore, these forests should be compared to our reference plots (Fig. 3.6).

The smaller mode at 25-29.9 cm compared to 40-49.9 cm found elsewhere could be explained by the very low productivity of sandhill soils on EAFB (Provencher et al. 1998). Nearly all trees measured here are found on Lakeland soils containing no more than 8% total silt and clay, whereas the soils underlying the Wade Tract (Red Hills) and the Escambia Experimental Forest have higher clay and silt content and productivity. We noted that the Wade Tract supports several trees >55 cm (Platt, Evans, and Rathbun 1988), which are exceptionally rare at EAFB (Fig. 3.6). Therefore, trees may just be smaller in diameter at EAFB. This statement, however, assumes that the cause of these modes at intermediate diameter classes is the same regionally. We do not know that this is the case, but a past record mast year may explain the bimodal longleaf pine size distributions. Platt, Evans, and Rathbun (1988), however, proposed that bimodality may be indicative of major catastrophic events, which they could not identify. They advanced this explanation because their theoretical analyses showed that the age and size distributions of longleaf pine at the Wade Tract were unstable, probably due to strongly disruptive abiotic factors.

The other discrepancy between our reference plots and the Wade Tract and the Escambia Experimental Forest was the lower number of longleaf pines in the smallest diameter classes at EAFB. It is this discrepancy that makes any burned restoration plot more similar to reference plots (Figs. 3.7-3.8). Our results, therefore, suggest that fire alone can cause this pattern on EAFB. The frequent maintenance fires in vegetation dominated by wiregrass (*Aristida beyrichiana*) at the Wade Tract and by bluestems at the Escambia Forest may be less harmful to saplings than the heavier fuel-driven fires experienced at EAFB. Provencher et al. (1998) showed that groundcover woody vegetation (<1.4 m tall) is more abundant than herbaceous vegetation at EAFB, and wiregrass is practically absent from most of the plots examined here. Heat generated by the combustion of mixed woody and herbaceous fuels may be more harmful to longleaf pine saplings than heat generated from the rapid combustion of wiregrass and herbaceous material.

The greater sensitivity of smaller diameter classes of longleaf pines to fire that we observed has also been reported by Streng et al. (1993) at St. Marks Wildlife Refuge, Florida, over a period of 7 years involving repeated burn treatments. They found that, in sandhill communities, longleaf pine mortality was 37% for trees of 2-4.9 cm DBH, 27% for trees 5-9.9 cm DBH, and <8% for all larger trees (Glitzenstein et al. 1995). Boyer (1990) reported greater longleaf pine mortality for the smallest (2.54 cm DBH class: 53%) and largest trees (56 cm DBH class: 25%) after July and August burns. Boyer (1990) observed that these burns caused greater damage to longleaf pines than to hardwoods after the first prescribed burn. At EAFB, the large amount of woody debris from felling hardwoods would be the most likely cause of more lethal fires for pines <10 cm DBH in felling/girdling plots compared to ULW® plots.

Oak and Longleaf Pine Juveniles

Juvenile oaks and longleaf pines showed opposite responses to fire. Oak juvenile densities generally increased after a burn or a felling operation (Fig. 3.9), due to resprouting (Waldrop et al. 1992, Rebertus et al. 1993). As expected, the only method that decreased oak juvenile densities was ULW®. Juvenile longleaf pine densities were invariably and equally decreased by fire in the growing and dormant seasons (Fig. 3.10). Mortality varied between approximately 50% and 60%. Other treatments had no effect on longleaf pine juveniles, which we expected since ULW® is not toxic to pines (McLemore 1983, Griswold 1984), and felling/girdling does not directly affect juveniles unless felled trees crush them. The percent mortality in burn plots was consistent with values reported in the literature (Boyer 1985, 1990, Grace and Platt 1995).

Longleaf Pine Seedlings

We reported median seedling densities between 37,000/ha and 50,000/ha for treatment plots and 100,000/ha for reference plots in 1996 (extrapolating from units in Fig. 3.11). These individuals were <1 year old. Dormant-season fuel reduction burns reduced these densities by approximately 95% after one year (Fig. 3.11). Background mortality was estimated at 50% from control and growing-season burn plots (not burned after 1995). Mortality in reference plots was 22%, despite seedlings having been exposed to prescribed burns and wildfires. Pre-grass stage seedling densities as high as 24,000/ha have been reduced to 2,600/ha within two years of growing season fire at the Wade Tract in southern

Georgia (Grace and Platt 1995), representing a 89% decrease. These results confirm the well-established fact that longleaf pine seedlings are readily killed by fire during their first year (Wahlenberg 1946). Also, control and burn plot results suggest that oak competition or the inability of seedling roots to contact mineral soil because of thick litter may inflict greater mortality on seedlings than fires in reference plots. We counted many seedlings growing on thick patches of leaf litter a few months after germination. We suspect, following field visits, that fires probably killed more seedlings at the Wade Tract because groundcover there is more dense and more continuous than in EAFB's reference plots. Graminoid cover is only 14% on average (Provencher et al. 1998) and litter is patchy within EAFB reference plots. Fine fuels are thus probably not sufficiently continuous for fire to reach everywhere on EAFB.

MANAGEMENT IMPLICATIONS

The trade-off between the time needed to reach a maintenance condition of the midstory (defined by the vertical structure of the tree component) and the cost of midstory reduction is clearly the main dilemma for managers. The issue of longleaf pine seedling and juvenile mortality should not overly constrain management, because all techniques investigated here appear to eventually affect longleaf pine juveniles equally through the repeated attrition of individuals by fire and other mortality factors. Moreover, we do not think that managers should focus on eliminating fire in the interest of saving a single mast year given that longleaf pines reproduce many times during their long life span and produce thousands more seedlings than can ever grow to merchantable size. Managers who need to rapidly remove the hardwood midstory due to endangered species mandates, for example, may consider techniques other than growing season burning, but at a potential cost to restoration of the overall system integrity, as defined by vegetation, insects, etc. The variation in oak mortality observed in growing season fire plots, however, suggests to us that repeated fires will easily reduce the hardwood midstory regardless of droughts and hurricanes. There are other good reasons to limit midstory reduction effects to burning only (Provencher et al. 1998; Chapter 4). Herbicide and mechanical hardwood reduction techniques will reduce oaks, but at a great financial cost. If financial cost is not a central issue and quick measures are needed, decisions should be based on whether hardwood resprouting following felling and the detrimental side effects of herbicides on the rest of the community are serious problems. Because the default management technique is to burn every 3-5 years at EAFB, resprouting does not seem to be a major decision factor because fire should control resprouts. In fact, oak resprouting may improve the habitat for invertebrate and bird species (Provencher et al. 1998). Effects of herbicides on the understory communities thus become an important element in the decision process of land managers. We cover this issue in Provencher et al. (1998) and in Chapter 4.

4. EFFECT OF HARDWOOD REDUCTION ON COMMUNITY SIMILARITY BETWEEN RESTORED, FIRE-SUPPRESSED AND REFERENCE LONGLEAF PINE SANDHILLS AT EGLIN AIR FORCE BASE, FLORIDA

ABSTRACT

We examined one of the most important themes of restoration ecology: using reference sites to measure restoration success while investigating restoration techniques. We report on one post-restoration study in degraded sandhills at Eglin Air Force Base, Florida, where restoration success was measured by indirect comparison to reference sites. Potential metrics of ecological condition were identified from the first three years after experimental units were treated. We compared the initial effects of three hardwood reduction techniques in fire-suppressed sandhills (growing season burn, ULW® form of the herbicide hexazinone, and mid-story mechanical felling/girdling) and a no-treatment control on the similarity between restoration and reference plots for: a) soil chemistry; b) tree densities and basal areas; c) different measures of understory vegetation; d) arthropod family and species/morphospecies densities; and e) the number of breeding bird species and their detection rates. For fall measurements taken in 1995-1997, restoration treatments were significantly more similar to reference plots than expected by chance in 7 of 27 groups of variables: basal area of all tree species in 1995 and 1997, groundcover plant density in 1997, the density of flowers per plant life form in 1995, vegetation cover groups in 1996 and 1997, and arthropod species/morphospecies density in 1996. For the spring measurements taken in 1996-1998, treatment effects on similarity to the reference condition were detected in 6 of 12 groups of variables: arthropod species/morphospecies densities in 1997, breeding bird detection rates from 1996 to 1998, and the number of breeding bird species (one variable group) in 1996 and 1997. Restoration treatments were not significantly more similar to the reference condition for soil chemistry, tree density, groundcover life form density, number of plant species, and herb-layer and soil/litter arthropod family density in any season or year. Growing season burning increased the similarity to the reference condition for arthropod species/morphospecies density and groundcover plant density. For groundcover plant densities, felling/girdling and ULW® plots subjected to fuel reduction fires were significantly less similar to the reference condition. ULW® application decreased the similarity to the reference condition for the density of flowers of plant life forms. In 1995 and 1997, fire increased percent cover group similarity by burning fine fuels. Treatments increased the similarity to the reference condition for breeding bird detection rates and species richness.

INTRODUCTION

The measurement of restoration success and selection of feasible metrics that track ecological change in terrestrial systems remains a poorly researched subject (Noss 1990, Hardesty, Gordon et al. 1997, Palmer et al. 1997). These methods are more developed in aquatic ecosystem and water quality control research (e.g., Karr 1991, Keddy et al. 1993, Barbour et al. 1996) than in upland systems. In the best of circumstances, restoration success of a degraded site would be measured by comparing the value of several variables (e.g., plant species densities) against the range of values from several representative and local sites with

high ecological integrity (reference sites) (Gordon et al. 1997, Palmer et al. 1997). This type of direct comparison to a reference site is generally not possible because reference sites, such as old growth forests, are rare. Even when reference sites are present, they may not be representative due to edaphic variation and unknown management histories (Rodgers and Provencher 1999). One solution to this problem was found in a study of biotic integrity for Florida streams, where Barbour et al. (1996) identified several reference streams by using multivariate techniques. They removed geologic variation from their selection process by grouping streams according to their biotas. Although the main goal of the study was to identify metrics of ecological condition, their approach could also be used to measure desired endpoints for stream restoration.

In the absence of any local reference site, published botanical accounts from other locations with similar soils (Clewell and Rieger 1997) and data from experiments may provide the best alternatives for setting restoration goals. Experimental treatments and management activities that imitate the dominant and natural processes (e.g., growing season fire) of a region can reveal the short- and long-term directionality of individual variables (e.g., forb density increases over time with periodic growing season fire). These quasi-natural treatments can then become a benchmark for other potentially more effective, but less natural experimental techniques of restoration (e.g., herbicides, mechanical midstory reduction).

Choosing metrics to track changing ecological conditions depends on human objectives related to these changes. Most management situations involve a variety of objectives and, in turn, these set the stage for the choice of metrics (Department of Defense-Air Force 1993). An important consideration for choosing metrics should be their statistical properties. Many threatened and endangered species are monitored by government agencies (e.g., U.S. Fish and Wildlife Service) and private groups (e.g., The Nature Conservancy) because of legal requirements or specific management objectives. Some of the defining traits of threatened and endangered species are their low numbers and patchy distributions. Thus, we would not expect parametric statistics to be appropriate for these species, because they will likely fail to meet the assumptions. One solution is to sample common species or variables that exhibit similar responses as the threatened and endangered species to natural perturbations or management activities, or that directly influence the species of concern. Herein lies a paradox: a species may be common precisely because it is capable of tolerating a broad range of conditions and is thus unlikely to respond to small disturbances or restoration activities. Finding sufficiently sensitive metrics among the many possibilities requires controlled perturbations of ecosystems as performed in experiments. These controlled studies should account for initial conditions that could possibly confound the effects of manipulations and include replication to allow estimation of within treatment variation.

We report on one post-restoration study in fire-suppressed longleaf pine (*Pinus palustris*) sandhills at Eglin Air Force Base (EAFB), Florida, where restoration success was measured by indirect comparison to reference sites and where potential metrics of ecological condition were identified from the first three years after experimental units were treated. We compared the initial effects of three hardwood reduction techniques in fire-suppressed sandhills (growing season burn, application of the ULW® form of the herbicide hexazinone, and mid-story mechanical felling/girdling) and a no-treatment control on the similarity between restoration and reference plots for: a) soil chemistry; b) tree densities and basal areas; c)

density, percent cover, and number of species of understory vegetation; d) arthropod families and species/morphospecies densities; and e) the number of breeding bird species and their detection rates. Because of the complexity of this study, the reader may want to consult the timeline of our project (Fig. 2.2) to help differentiate sampling phases.

METHODS

Study Area

EAFB occupies the southern portions of Walton, Okaloosa, and Santa Rosa Counties in the western Florida Panhandle (Fig. 3.1). The climate is temperate with mild winters and hot, humid summers. A more detailed description of weather, soils, and general botany is presented in Provencher et al. (1998) and in Rodgers and Provencher (1999).

Experimental Design

Restoration Blocks. The experimental design and choice of restoration plots and reference plots are described in Provencher et al. (1998) and Chapter 3. We briefly present them here. The four treatments were growing season burning, application of the ULW® form of the herbicide hexazinone (75% active ingredient applied at a rate of 2.44 kg/ha [2 lb/acre]), chainsaw felling/girdling, and a control (maintenance of fire-suppression). One replicate of each treatment was randomly assigned to a 81-ha (200-acre) plot in one of six blocks in a randomized complete split-plot block design (Steel and Torrie 1980) (Fig. 3.1: B-7, Wolf Creek, Meets Creek, Malone Creek, Exline Creek, and C-72). Following pre-treatment sampling from spring 1994 to spring 1995, most treatments were applied in April, May, and June 1995. Felling/girdling operations commenced in June and proceeded until late August in the 20-ha (50-acre) sampling areas of each plot. The majority of the remaining 61-ha (150-acre) portion of the plots was felled/girdled by August, but operations persisted until November in two plots. ULW® and felling/girdling plots were burned for fuel reduction from early March to late April in 1997.

In each 81-ha plot, all subplots and sampling stations were located in the 20-ha corner farthest from the neighboring plots of the block in order to minimize inter-treatment effects (Fig. 4.1). We borrowed from split-plot terminology to label our nested sampling units: each plot contained 32, 10×40-m subplots (Fig. 4.1); any sampling unit within a subplot was referred to as a sub-subplot.

Reference Blocks. A total of six 81-ha frequently burned, longleaf pine (*Pinus palustris*)-dominated sandhill plots were established to represent objective goals for the restoration of fire-suppressed plots (Fig. 3.1: A-77, A-78, and B-75). Reference plots, although not part of the restoration experimental design, provide a benchmark for measurement of the success and efficacy of the restoration treatments applied. Each reference plot contained the same subplot sampling design as the restoration plots (Fig. 4.1), but the 20-ha sampling site was located in the plot centers to avoid potential edge effects on these sites.

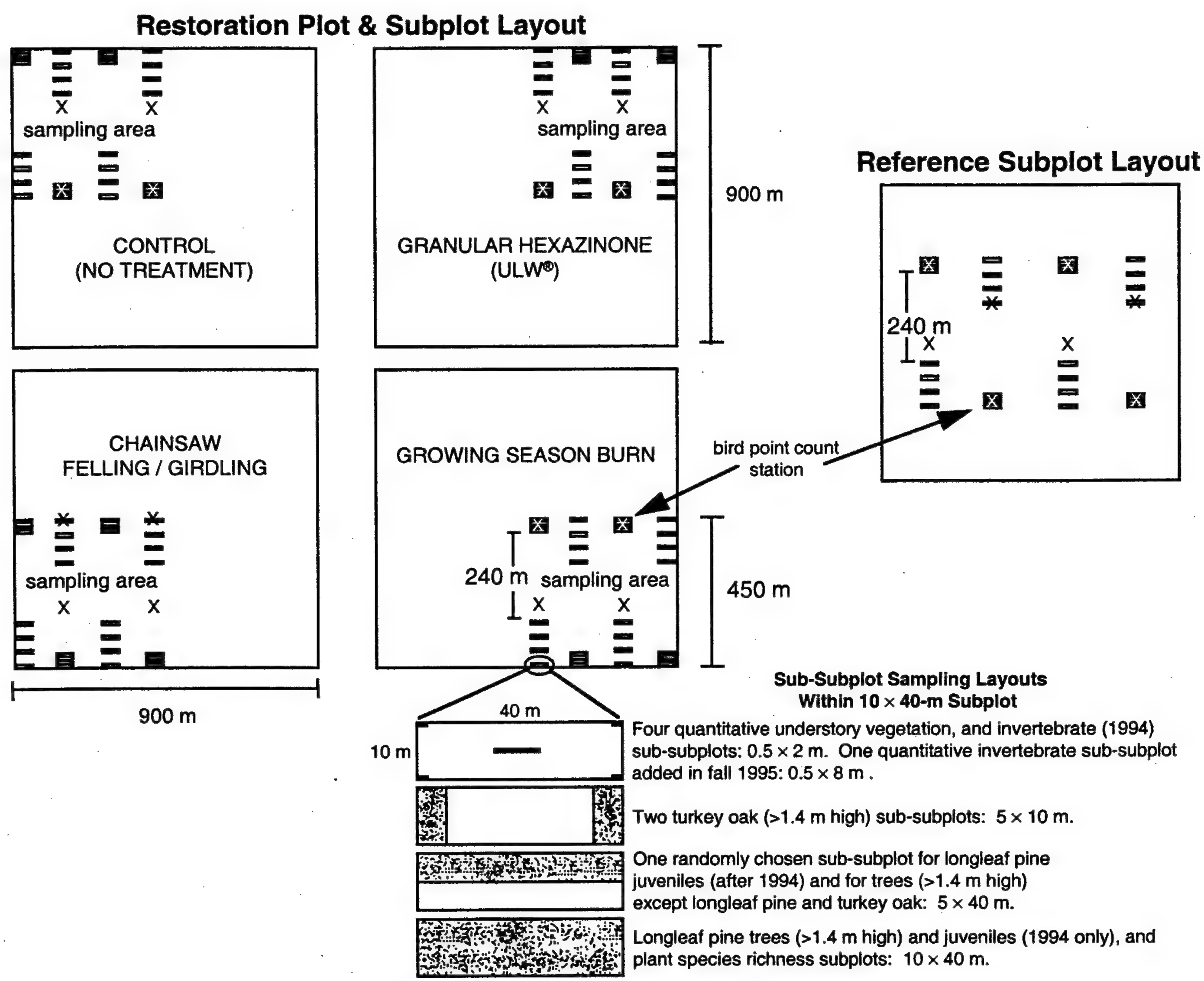

Fig. 4.1. Sample layout of 81-ha (200-acre) in one of six reference plots and in restoration plots and sampling areas in one of six blocks in a randomized complete block split-plot design consisting of four whole-plot treatments. Spatial randomization of treatments varies per block.

Data Collection

We present the various pre-treatment sampling periods for birds, soils, vegetation, and arthropods in chronological order (see Fig. 2.2 for the timeline). We sampled spring pre-treatment breeding bird detection rates from 4 May to 30 June 1994. Fall pre-treatment soil chemistry cores, vegetation density and cover groups, and herb-layer arthropod densities were collected from 15 July until early December 1994. Fall/winter pre-treatment tree densities, DBH, and height were sampled from 1 November 1994 until 1 April 1995. Spring pre-treatment sampling for soil chemistry cores, herb-layer arthropods, and soil/litter arthropod detection rates was conducted from 1 April until 15 June 1995. (Groundcover vegetation was also sampled, but we do not present the results in this report.) Restoration treatments were applied in the spring and early summer of 1995. (Plots were sampled just prior to application.) Treatment application on these plots marked the end of the pre-treatment phase of the study.

Post-treatment sampling periods approximately matched pre-treatment dates, with the following exceptions. Breeding birds were sampled only in reference plots in 1995 because treatment application interfered with bird activity and detection. Also, breeding bird sampling started slightly later (14 May) in 1996 due to logistical problems. Due to methodological constraints, a spring pre-treatment collection of soil cores was not performed—we substituted the fall 1994 data for the pre-treatment sample. Soil/litter arthropods were not sampled after spring 1996. Trees were not sampled during the fall 1996/winter 1997. Fall arthropod sampling was conducted from 15 July to early October after 1994.

Tree Species. Protocol used in tree sampling is presented in detail in Chapter 3 of this report. All trees >1.4 m high were measured and longleaf pine seedlings and juveniles were counted.

Soil Chemistry. Four soil cores (30-cm deep) were collected from the corners of each 10 × 40-m subplot. The four cores were mixed to create a single sample, resulting in 32 samples for each 81-ha plot. Each subplot sample was analyzed separately (Soil Testing Laboratory, University of Florida) for Al, Ca, Cl, Fe, K, Mg, Na, P, pH, total Kjeldahl nitrogen (TKN), and organic matter. The Mehlich-1 extraction procedure was used for Ca, Mg, P, K, Na, Al, and Fe. Walkley-Black dichromate methodology was used for organic matter determination. TKN was determined using the micro-Kjeldahl method.

Groundcover Vegetation Densities. Groundcover vegetation densities (including tree species <1.4 m high) were estimated in all restoration and reference plots. Densities were estimated by counting individual plants or stems in the four 0.5 × 2-m corner sub-subplots of each of the 32 subplots per 81-ha (200-acre) plot (Fig. 4.1) in which percent cover was also estimated. All plants <1.4 m high and rooted within each sub-subplot were recorded; 50% of a clump had to be rooted within the sub-subplot to be counted. Highly abundant species—Darrow's blueberry (*Vaccinium darrowii*), dwarf huckleberry (*Gaylussacia dumosa*), gopher apple (*Licania michauxii*), grass-leaf golden aster (*Pityopsis graminifolia*), and pineland hoary-pea (*Tephrosia mohrii*)—were assigned to density classes: I = 1-5; II = 6-10; III = 11-25; IV = 26-50; V = 51-100; VI = 101-150; and VII = >151 stems. For graminoids, clumps separated by >10 cm were considered distinct plants. For all species, the

number of flowering stems or clumps was also recorded. A 10-minute "walk-through" of the 10 × 40-m plot was conducted to identify plant species present. Plant identifications were made based mainly on Clewell (1985).

We organized groundcover vegetation densities into eight plant life forms. These life forms represent nearly all plant species in our plots. The selected plant life forms and their distinguishing features are described following Mueller-Dømbøis and Ellenberg (1974). Graminoids include all grasses and sedges, including wiregrass and Gray's beakrush (*Rhynchospora grayi*). Non-leguminous forbs (hereafter termed "forbs") are non-woody plants other than graminoids or rushes; yellow buttons (*Balduina angustifolia*) is an example. Legumes are also forbs, but were analyzed separately, and refer to those species in the Fabaceae, a group of plants bearing dry, single-celled fruits that open along two sutures, e.g., Florida milk pea (*Galactia floridana*). Clonal shrubs are woody plants that reproduce vegetatively by sending rhizome or root sprouts, giving the appearance of numerous shrubs (Godfrey 1988); gopher apple is a common example. Non-clonal shrubs refer to woody plants with multiple stems arising from a common root stock (hereafter termed "shrubs") that are <2 m tall, e.g., woody goldenrod (*Chrysoma pauciflosculosa*). Woody vines grow by supporting themselves on other plants or trail along the ground and include catbrier (*Smilax auriculata*). Small trees, such as winged sumac (*Rhus copallina*), are 2-5 m tall. Large trees are 5-50 m tall when mature (we measured them in the groundcover) and include longleaf pine.

Understory Cover Groups. Cover was estimated in four 0.5 × 2-m sub-subplots (Fig. 4.1) for graminoids, wiregrass and pineywoods dropseed, forbs, lichens, woody species, bare ground, fine litter, woody litter, and cryptobiotic crust (black form). Graminoids were defined as all grasses and sedges except wiregrass and pineywoods dropseed. This separation of wiregrass and pineywoods dropseed from graminoids was done to measure the finer fuel component among several other fine fuels. Forb cover included legumes. Percent cover was estimated in seven cover classes: 0 = 0%; 1 = 1-5%; 2 = 6-25%; 3 = 26-50%; 4 = 51-75%; 5 = 76-95%; and 6 = 96-100%. The following criteria were used in estimating cover: 1) bare ground counted if > 1 cm^2 of mineral soil was visible; 2) lichen cover included only lichens that grew directly on the ground; 3) woody litter included wood covered with lichen; 4) soil with fine root mat and fine organic matter was counted as fine litter; 5) any woody vegetation (<1 m high) was included in cover estimates; 6) plants included in cover were not necessarily rooted within the sub-subplot; 7) percent leaf area of compound leaves was estimated as 60% of the contoured leaf area. Cover of the vegetation could overlap that of soil and each other (e.g., woody species leaf area over grasses) so total cover in the sub-subplots could exceed 100%. Percent cover of the combined tree midstory and canopy (hereafter referred to as "canopy cover") was measured at both 0.5-m ends of each sub-subplot using a spherical densiometer. We reported cover as a proportion in the figures (e.g., 50% cover is reported as 0.5).

Soil/Litter Arthropod Sampling. Using a 15-cm diameter cylinder, we collected one litter sample, including the top 2-cm of the soil/litter matrix, along a randomly selected edge of each 10 × 40-m subplot (i.e., 32 samples for each 81-ha plot). Invertebrates were isolated by live heat extraction for 48 hr. The resulting samples were then processed with heptane. Soil/litter invertebrate densities were estimated by family, superfamily, or order.

Herb-Layer Arthropod Sampling. Herb-layer arthropod densities were estimated by family, superfamily, or order in restoration and reference blocks. In some cases, specimens were identified to species or morphospecies (hereafter named morpho/species). Morphospecies are taxa that can be readily separated by non-specialists using obvious morphological differences (Oliver and Beattie 1996). To successfully collect arthropods of various sizes and mobility, individuals were first collected using a sweep net, followed immediately by a modified D-Vac insect vacuum. In 1994, individuals were collected from herb-strata vegetation (<1.4 m) within the same four 0.5 × 2-m areas used for understory vegetation sampling (Fig. 4.1). Because we suspected that the noise and motion of our vacuum device and sweep net were flushing some arthropods from adjacent sub-subplots at the 10-m sampling distance, we changed the location and shape of arthropod sampling sub-subplots to a single 0.5 × 8-m rectangle situated in the center of the subplot beginning in fall 1995 (Fig. 4.1). Moreover, the second method minimized escapes by arthropods because we opened the sweep net once instead of four times.

We selected 6 orders and 33 families of insects and spiders to target for species and morpho/species analysis. These selected orders and families contained morpho/species that occurred in high density, were collected mostly as adults, and remained identifiable after D-Vac collection. We enlisted the assistance of taxonomic specialists where possible to perform initial species identifications and established a reference collection of 288 authoritatively identified adult arthropod morpho/species (Provencher et al. 1998). This growing collection has permitted us to determine the density of some morpho/species. However, most immature specimens of these and other species cannot be easily identified below the family level. For example, few grasshoppers, which are abundant in sandhills, could be identified to the species level, because most individuals captured in the spring were early instar nymphs.

Breeding Bird Detection Rates. Plots were surveyed using a variation of the unlimited distance point count method (Blondel et al. 1981). In restoration plots, point counts were conducted from four permanent stations placed uniformly (approximately 200 m apart) within the 20-ha (50-acre) sampling corner of each plot (Fig. 4.1). Because detectability decreases with distance, this station arrangement alleviates potential for recording individuals that may actually be within adjacent plots of a given block. In reference plots, point counts were conducted from eight stations uniformly located approximately 200 m apart in plot centers (Fig. 4.1). The difference in station arrangement between restoration and reference plots reflects restrictions imposed by other aspects of the study design and should not impact data collected.

Each sampling day, one block (i.e., four restoration or two reference plots) was surveyed for birds. Counts commenced when daylight reached a minimum level necessary to visually identify birds. For each 8-minute point count, the species, number, and location of all birds seen or heard in the plot were recorded. Birds flying over the plot were recorded only if there was evidence that they were foraging over the plot (e.g., American kestrel [*Falco sparverius*]) or had been perched in the plot prior to taking flight. Two observers surveyed an entire block (16 total stations) in approximately 3 hours. Thus, at a rate of 1 block/day, one full round of counts in which all nine restoration and reference blocks were surveyed was completed in 9 days. This rate allowed a maximum of 3-5 rounds to be conducted during the spring, dependent upon weather and military mission activity.

The survey order for each block within a given round of bird surveys was determined randomly, but may have been modified due to military operations. For spring 1994 and 1996, survey order of individual restoration plots on a given day was determined randomly. A further restriction was that, by round #4, each of the four plots in the block would be sampled 1st, 2nd, 3rd, and 4th in order only once during a given day. For reference blocks, survey order of individual plots on a given day was similarly randomized, except that each of the two plots comprising these blocks was to be sampled 1st and 2nd in order exactly twice by round #4. These measures attempted to minimize temporal bias in survey data by accounting for the tendency of bird activity, and therefore detectability potential, to decline throughout the morning and to vary from day to day. This sampling arrangement was changed during spring 1997 to produce a more systematic, but random order of restoration plot survey by treatment type.

Purging Mesic Reference Subplots. We purged reference subplots that were not representative of the sandhill restoration subplots. We used canonical correspondence analysis (Kenkel and Orlóci 1986, Ter Braak 1986) and non-metric multidimensional scaling (Kruskall 1964, Kenkel and Orlóci 1986) on soil texture variables (i.e., percent total silt and clay and percent sand of different grain sizes) to identify these subplots as outliers (Provencher et al. 1998). Unfortunately, because percent total silt and clay was not the best predictor of subplot ordination, we could not remove subplots that were clearly akin to flatwoods due to underground water flow and very high silt and clay content. We found that the simple ranking of subplots by percent total silt and clay identified mesic subplots often associated with underground sources of water or proximity to creeks and depressions. All reference subplots with >8% total silt and clay were discarded, affecting only plots A-78 west and east.

Similarity Calculations

Similarity measures allowed us to directly determine the treatment that most increased the resemblance between the restored and reference plots. We calculated similarity of selected variables between each treatment plot and each reference site using two formulas, although there is a wide variety of indices available (Brower et al. 1989, Underwood and Chapman 1998). Proportional similarity was calculated for all multivariate data sets, except soil chemistry variables, because it is widely used in ecology and takes into account the relative abundance of variables. This last feature was especially important to us because site conditions (e.g., soil productivity) may greatly change the abundance of variables (e.g., insect densities) but not their relative abundance in a site. We were not interested in detecting site differences. For groups of variables with non-negative quantities (e.g., groundcover vegetation species densities), proportional similarity (PS) between each treatment plot i (= 1,..., 24) and each reference plot j (= 1,...,6) was calculated as

$$PS_{ij} = 1-0.5\sum_{k=1}|p_{ik}-p_{jk}| \qquad \text{(Brower et al. 1989)},$$

where p_{ik} is the proportion of the logarithm of variable k in treatment plot i and p_{jk} is the proportion of the logarithm of variable k in reference plot j. We took the logarithm of variables to prevent high magnitude values (e.g., abundant species) from dominating PS and, thus, to increase the representation of rarer species or uncommon variables. Plots that share all the same variables in the same proportions will have a PS = 1, whereas plots that share no variables will have a PS = 0.

The above formula was averaged over all reference plots per restoration plot i with a sum weighted by sample sizes,

$$PS_i = \sum n_j PS_{ij}/N,$$

where n_j is the number of subplots in reference plot j and N is the total number of subplots in all reference plots.

We created a second index of similarity for two reasons: a) we needed a formula bound between 0 and 1 that would accommodate negative values (see below why this applies to soil chemical variables) and single variables (number of species and canopy cover); and b) we wanted to incorporate within-plot variation from the restored and reference plots into the measure of similarity. Taking the negative exponential of the absolute value of the *t* statistic would achieve these goals and be somewhat recognized from the normal probability distribution function (Sokal and Rohlf 1981). Therefore, for single variables or for groups of variables with positive and negative values (see below), we calculated the weighted average similarity by a formula, termed endpoint difference (ED),

$$ED_i = \sum_{j=1} n_j \sum_{k=1} \exp[-|Z_{ik} - Z_{jk}|/\sigma_{eij}]/K/N,$$

where the value within the exponential function is the absolute value of the *t* statistic (Sokal and Rohlf 1981), σ_{eij} is the joint standard error of Z_{ik} and Z_{jk} assuming unequal sample sizes, K is the number of variables, and Z_{ik} is either the average of a single variable k from plot i (similarly for plot j) or the average Z-transformed (Sokal and Rohlf 1981) value of variable X_{ik}. Z-transformation was calculated as,

$$Z_{ikm} = (X_{ikm} - \text{average of } X_k)/\text{standard deviation of } X_k,$$

where Z_{ikm} represents the m^{th} transformed value of X_k in plot i ($X_{ik1}, \ldots, X_{ikT}$) (similarly for plot j), T is total number of subplots in all 24 restoration plots (independent calculation for all 6 reference plots), and the average and standard error of X_k are calculated over all subplots from all restoration (or all reference) plots. To obtain Z_{ik} as in the ED index, all Z_{ikm} were averaged per plot.

We Z-transformed soil chemistry values to standardize variables having different units (e.g., pH and ppm) and widely different magnitudes. Z-transforming variables can produce positive and negative values, which are incompatible with proportional similarity. Plots that share all the same variables in the same proportions (all $|Z_{ik} - Z_{jk}| = 0$) or that have large variability preventing these plots (standard errors are large) from being differentiated, respectively, will have an ED equal or close to 1. Plots that share no variables (all $|Z_{ik} - Z_{jk}|$ are large) or if samples can be easily distinguished due to small variability (standard errors approach 0) will have an ED close to 0.

Statistical Analyses

We tested restoration treatment effects on similarity with a randomized complete block analysis of covariance (two-way ANCOVA) (Steel and Torrie 1980). The subplot level (sampling distance) of the split-plot design was not tested. We tested the effect of pre-treatment data on post-treatment data using ANCOVA. Pre-treatment data were used to adjust post-treatment averages to account for differences among treatments that existed prior to treatment application. The adjusted averages were used to rank similarities among treatments. We performed ANCOVAs using a computer randomization test (Edgington 1987). The details of the randomization procedure are described in Provencher et al. (1998).

Similarity (a proportion) was arcsine-square-root transformed to stabilize the variance (Steel and Torrie 1980).

When significant treatment effects were detected for similarity, we used the above ANCOVA methodology to test treatment effects on the variables that most contributed to the similarity index. For proportional similarity, the contribution of each variable per treatment was arbitrarily measured by $1-\sum_{v=1}\sum_{j=1}|p_{vk}-p_{jk}|$, where the sums were calculated over the six reference sites and the six replicates per treatment and per variable. Similar summations were performed for the endpoint difference: $\sum_{v=1}\sum_{j=1}\exp[-|Z_{vk}-Z_{jk}|/\sigma_{evj}]$. The variables with the strongest contributions were identified by two intuitive criteria: a) the largest standard deviations (i.e., calculated from the four average contributions) among treatments and b) the rank of average contributions among treatments matched as closely as possible the rank of average similarity among treatments. We corroborated that both criteria resulted in a choice of variables that closely resembled findings obtained by canonical correspondence analysis (Ter Braak 1986).

We contrasted the following treatments (see Chapter 3 in this report for more details): control versus growing season burn (C vs. B), burn versus ULW® (B vs. U), and ULW® versus felling/girdling (U vs. F). In the first contrast, we tested whether maintaining fire suppression (control) was as efficient as burning. Burning is the management default at EAFB, because it is the least expensive management tool and because chronic fires would characterize the maintenance condition of sandhills. Because felling/girdling and ULW® are more expensive management techniques than burning, their effects were compared to burning, but not directly to fire suppression (control). These more expensive methods were expected to more effectively reduce hardwood abundance than burning. We compared burning to ULW® to contrast techniques that differ greatly in the amount of oak (*Quercus* spp.) resprouting and topkill mortality they cause. Burning should result in approximately 50% topkill and abundant resprouting (Glitzenstein et al. 1995), while ULW® should practically eliminate oaks. ULW® was compared to felling/girdling because of a management need to discriminate between equally expensive methods that differ in oak resprouting: resprouting is stimulated by felling oaks.

We predicted that growing season fire would cause soil chemistry, groundcover plant, tree, arthropod, and bird species/variable assemblages to become more similar to the reference condition, which is frequently burned, than the control plots. The effects of ULW® on the similarity of the understory to the reference condition may be more difficult to predict than those of burning. By killing (complete kill) oaks, soil moisture available to plants should increase in ULW® plots, and as a result, plant growth in the form of graminoid and forb cover should increase (Brockway et al. 1998). But, because ULW® also should cause leaf litter to increase and woody species and bare ground to decrease, the contributions of cover groups may cancel out their effects and, thus be no more similar to the reference condition than the control plots. This herbicide also should kill non-target plants (Provencher et al. 1998), thus reducing the similarity between ULW® and reference plots for plant density. We expected a slow response of the vegetation to felling/girdling because of increased sunlight to the groundcover and its low impact nature. Arthropod densities will increase more rapidly in the presence of new vegetation growth, which should be more pronounced with fire and, to some extent, with felling operations due to resprouting and increased light to the herb layer. We predict that arthropod assemblages will be more similar to the reference

condition in these two treatments than in the control and felling/girdling plots. Bird species assemblages, which are known to respond most to the structure of their habitat, are predicted to be more similar to the reference condition in ULW® and felling/girdling plots than in burn plots, which will also be more similar to the reference sites than the control plots.

RESULTS AND DISCUSSION

We detected significant effects of restoration treatments on the similarity between treatment and reference plots in 7 of 27 variables or groups of variables measured each fall from 1995 to 1997 (Table 4.1a). From the spring measurements of 1996 to 1998, significant treatment effects on similarity were detected in 6 of 12 variable groups or single variables. When a significant effect was detected, variables that contributed more strongly to the index of similarity and to the ranking of treatments are discussed. We retained between 2 to 8 variables depending on taxa and magnitudes of contributions to similarity.

Fall Similarity

Trees. In 1995 and 1997, basal area was significantly more similar to the reference condition in ULW® and felling/girdling treatments than in the burn treatment, which had significantly higher similarity than the control treatment (Table 4.1a). Longleaf pine and turkey oak (*Quercus laevis*) contributed most to the difference in similarity values among treatments (Table 4.1b). Because the basal area of longleaf pine in reference plots was much higher than that of any treatment, small and non-significant treatment differences among restoration plots were magnified in similarity calculations (Provencher et al. 1998, Table 4.1b). We suspect that a few replicate control plots coincidentally contributed least to similarity, although this is not reflected in basal area differences among treatments in Table 4.1b. The case of turkey oak was more revealing, because the amount of basal area removed by restoration treatments (Chapter 3) actually explained similarity patterns (Table 4.1b).

We had expected that if basal area similarity yielded significant treatment effects, then tree density and canopy cover would show the same significant results. This expectation was not met. The case of canopy cover is explained by greatly overlapping distributions between restoration and reference plots (Provencher et al. 1998). As a result, the joint standard error produced by the overlap would substantially decrease the endpoint difference index. The case of tree density is more difficult and subtle to elucidate, because oak densities were dramatically reduced by felling/girdling and ULW® treatments (Provencher et al. 1998, Chapter 3). The two main causes of this odd result may be non-oak hardwoods and differences between longleaf pine basal area and density. First, common non-oak hardwood densities, such as sparkleberry (*Vaccinium arboreum*) and yaupon (*Ilex vomitoria*), were not significantly affected by treatments (Provencher et al. 1998) and therefore, they diluted the effect of oak density on proportional similarity. These non-oak hardwoods contributed very little to basal area due to their small sizes. Second, small differences in longleaf pine densities between restoration and reference plots, but large differences in basal areas (Provencher et al. 1998), may have diminished the contribution of longleaf pine density to similarity while accentuating the contribution of basal area.

Groundcover Vegetation. In 1997 only, the similarity of groundcover vegetation density to the reference condition was significantly higher in burn plots than in all other plots (Table 4.1a). The variation in similarity among treatments was most influenced by three woody

species: dwarf huckleberry, longleaf pine juveniles and seedlings (counted as part of the vegetation sample, not the longleaf pine juvenile winter count), and gopher apple (Table 4.1b).

The delayed response of groundcover plant similarity to treatments is interesting, because it follows the dormant season fuel reduction burns in ULW® and felling/girdling plots. We suggest that decreased similarity in ULW® and felling/girdling plots was attributed to reduction of dwarf huckleberry and longleaf pine juvenile and seedling densities by these fires (Table 4.1b, Chapter 3). The negative effect of hot ground fires on huckleberry species has been documented in the New Jersey Pine Barrens (Matlack et al. 1993). Dwarf huckleberry abundance and cover have been shown to peak one year after fire in flatwood scrub in central Florida (Abrahamson 1984), but to not respond after dormant season fires of any frequency compared to no-burning in southern Georgia flatwoods (Brockway and Lewis 1997). Dwarf huckleberry was not significantly reduced by growing season fire compared to no fire in 1995 (Provencher et al. 1998). Longleaf pine seedlings from the mast crop of 1996 were decreased by 95% by fuel reduction fires, and juveniles showed decreases of 50% (Chapter 3). Gopher apple densities were adversely affected by ULW® application compared to other treatments prior to fuel reduction burns (Provencher et al. 1998). Because adjusted gopher apple densities were highest in felling/girdling plots (Table 4.1b), which received the most intense fuel reduction fires (Chapter 3), we believe that gopher apple densities have not recovered from the initial ULW® application, which also received fuel reduction fires.

For the densities of flowers from all life forms, similarity was affected by treatments only in 1995. Felling/girdling, burn, and control plots were significantly more similar to reference plots than were ULW® plots (Table 4.1a). Graminoids and forbs, in decreasing order of importance, influenced most the significance of the treatment effect (Table 4.1b). The significant variation in life form flower density similarity was clearly determined by the negative effect of ULW® application on graminoids and forbs (Table 4.1b). We had expected that fire would stimulate flowering in burn and reference plots (Platt, Evans, and Davis 1988), but we observed that growing season burns or dormant season fuel reduction burns had no effect on similarity patterns for life form flowers. In the first year post-treatment we determined that ULW® application significantly decreased the density and the flower density of non-legume forbs, which were more abundant in felling/girdling and reference plots (Table 4.1b). Although graminoid flower density did not significantly vary among treatments in 1995 due to patchy numbers, the average density of flowers in reference plots was intermediate between burn and ULW® plot densities (Table 4.1b). Because of high variance, we believe that graminoid flowers would not have contributed to similarity had we used endpoint difference in place of proportional similarity.

The similarity of cover groups was significantly affected by restoration techniques in 1996 and 1997 (Table 4.1a). In 1996, burn plots were significantly more similar to the reference condition than all other treatments. Variation in cover of bare ground and fine litter contributed most to the significance of the treatment effect (Table 4.1b). The ranking of treatments for fine litter changed in 1997 compared to 1995, because both ULW® and burn plots were more similar to the reference condition than the felling/girdling and control plots (Table 4.1a). Fine litter and lichen cover were the dominant contributors to the significant result (Table 4.1b). The reversal in ranking of treatments for percent cover similarity from 1996 to 1997 was caused by relative changes in fuels before and after fuel reduction burns,

Table 4.1a. Effects of hardwood reduction techniques on the similarity between restoration and reference plots for trees, soil chemistry, groundcover, and arthropod variables from fall 1995 to 1997 (post-treatment). Adjusted values from each treatment were ranked from highest to lowest. Higher rank indicates a greater similarity of treatment plots to reference plots for the examined variable. Inequality signs are only presented for significant contrasts. The "?" sign indicates an uncertain outcome for an untested contrast. Pre-treatment effects were factored out of these summary results. Similarity was measured by proportional similarity for all multivariate groups (i.e., all variables except canopy cover and number of plant species) and by the endpoint difference index for univariate variables (canopy cover and number of plant species). Soil chemistry was normalized with the Z-transformation and similarity values calculated with the endpoint difference index.

Variable	Highest ↔ Lowest	Highest ↔ Lowest	Highest ↔ Lowest
	Fall 1995	Fall 1996	Fall 1997
Trees			
Density	U = F = B = C†	...‡	F = U = B = C
Basal area	U = F > B > C	...‡	U = F > B > C
Canopy cover	F = B = U = C	U = B = F = C	B = C = F = U
Soil chemistry	...§	U = C = B = F	U = C = B = F
Groundcover vegetation			
Density	C = U = B = F	B = C = F = U	B > C ? U = F
Life form density	B = C = U = F	F = U = B = C	B = F = C = U
Life form flower density	F = B = C > U	C = B = F = U	C = F = B = U
Cover groups	B = C = F = U	B > C = F = U	U = B > F ? C
No. species	B = C = F = U	C = B = F = U	C = U = B = F
Arthropods			
Family density	U = B = F = C	...§	...§
Morpho/species density	B = F = U = C	B > F = U = C	...#

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.

‡ Trees not sampled during the winter 1996/1997.

§ Data not available in this format.

Data processing in progress.

Table 4.1b. Variables most likely to influence similarity patterns found significant for restoration effects on trees, groundcover, and arthropod variables from fall 1995 to 1997 (post-treatment). Variables were ranked from contributing most to least weight to the similarity index value. Within variables, adjusted averages from each treatment were ranked from highest to lowest. Higher rank indicates a greater similarity of treatment plots to references plots for the examined variable. Inequality signs are only presented for significant contrasts. The "?" sign indicates an uncertain outcome for an untested contrast. "(R)" indicates that the position of reference plots is untested. See Table 4.1a for statistical analysis.

Variable	Most Highest ←↕→ Lowest Least Fall 1995	Most Highest ←↕→ Lowest Least Fall 1996	Most Highest ←↕→ Lowest Least Fall 1997
Trees			
Basal area	Longleaf pine (R) U = C = F = B† Turkey oak C > B > U = (R) F		Longleaf pine (R) U = C = F = B Turkey oak C > B > F = (R) U
Groundcover vegetation			
Density			Dwarf huckleberry C = B = F (R) > U Longleaf pine<1.4 m (R) B = C > F = U Gopher apple F > B = C (R) > U
Life form flower density	Graminoids B = F = C = (R) U Forbs (R) F ? C = B > U		
Cover groups		Bare ground (R) B > C = F > U Fine litter U > C = F > B (R)	Fine litter C > B (R) > U = F Lichen C > B = F = U (R)
Arthropods			
Morpho/species density		*Erythroneura* spp. (R) B = C > F = U *Mangora gibberosa* U = C = F = B (R) *Sminthurus* sp. #1 (R) B > U = C = F *Pseudomyrmex pallidus* C = U = F = B (R) *Thiodina* sp. #1 C = F = B = U (R)	

† Abbreviations: B= burn; C = control; F = felling; R = reference; U = ULW® herbicide.

demonstrating the importance of bare ground and fine litter. Fire reduced fine litter and lichens (fire intolerant), creating a condition similar to the reference sites characterized by frequent fire (Provencher et al. 1998). Therefore, plots that burned the most recently and hottest most resembled the reference condition.

Arthropods. In 1996, morpho/species densities were significantly more similar to the reference condition in burn plots compared to all other treatments (Table 4.1a). In decreasing order of importance, five morpho/species most explained the significant effect: *Erythroneura* spp. (Homoptera: Cicadellidae), *Mangora gibberosa* (Araneae: Araneidae), *Sminthurus* sp. #1 (Collembola: Sminthuridae), *Pseudomyrmex pallidus* (Hymenoptera: Formicidae), and *Thiodina* sp. #1 (Araneae: Salticidae). Two herbivorous species, *Erythroneura* spp. and *Sminthurus* sp. #1, were more abundant in the reference and burn plots (Table 4.1b). We hypothesize that these species experienced stronger population growth due to the more palatable forage with higher nitrogen content found after fires (Smith and Young 1959, Owensby et al. 1970, Nagel 1973, Harris and Whitcomb 1974, Dunwiddie 1991, Stein et al. 1992), and that fire may have increased the density of host plants and, thus, the density of these herbivores. Provencher et al. (1998) reported this same effect for spring arthropods at EAFB. The morpho/species *M. gibberosa* and *Thiodina* sp. #1 were least abundant, or in the case of *P. pallidus,* absent in the reference and burn plots, although these differences were not significant (Table 4.1b). These latter species are not herbivorous. In fact, *Mangora* and *Thiodina* are spiders that may be killed by fires. Fire had negative effects on spiders compared to a no-burn plot in a pristine Kansas prairie (Nagel 1973). *P. pallidus* was a patchily-distributed infrequent ant in our plots, and we should not attribute too much importance to its contribution.

Spring Similarity

Arthropods. In 1997, morpho/species density similarity was significantly greater in burn plots compared to all other treatments (Table 4.2a). Eight morpho/species contributed most to the significance of the restoration effect: *Sminthurus* sp. #1, *Stilpon limitaris* (Diptera: Empididae), *Attalus* sp. #2 (Coleoptera: Melyridae), *Pheidole floridana* (Hymenoptera: Formicidae), *Metcalfa pruinosa* (Homoptera: Flatidae), *Leptothorax pergandei* (Hymenoptera: Formicidae), *Erythroneura* spp. and *Crematogaster ashmeadi* (Hymenoptera: Formicidae) (Table 4.2b). *Sminthurus* sp. #1, previously identified as *S. carolinensis* (Provencher et al. 1998), and *Erythroneura* spp. were also major contributors to similarity in the fall 1996 (Table 4.1b). *Sminthurus* sp. #1, *Stilpon limitaris*, *Metcalfa pruinosa*, and *Erythroneura* spp. were identified as significantly responding to restoration effects, especially the burn treatment, in spring 1996 (Provencher et al. 1998). We found that these species continued to respond to the burn treatment in 1997 (Table 4.2b). Of these four morpho/species, only *S. limitaris* is not herbivorous (Cumming and Cooper 1992), and it was least abundant in control, ULW®, and growing season burn plots (Table 4.2b). The remaining morpho/species were three ants (*P. floridana*, *L. pergandei*, and *C. ashmeadi*) and one beetle (*Attalus* sp. #2), all of which incrementally contributed to the similarity calculation because of lower densities in burn and reference plots (only significantly so for *L. pergandei*) (Table 4.2b). The ants may be either a pollen feeder or general feeders with a wide range of food preferences (sweets, scavenging dead insects, etc.). *Attalus* sp. #2 may be a predator that waits on foliage for prey or a pollen feeder (White 1983). Therefore, these non-herbivorous species may benefit only indirectly from regenerating foliage. James et al.

(1997) have recognized the arboreal ant *C. ashmeadi*—commonly found on longleaf pines—as an important source of food for red-cockaded woodpeckers (*Picoides borealis*) in the Apalachicola National Forest (ANF), Florida. However, in the ANF, *C. ashmeadi* was found in higher densities in poorly burned areas, but the diversity of other species of arboreal ants, also red-cockaded woodpecker food sources, was higher in well-burned areas (James et al. 1997).

Birds. From 1996 to 1998, the ranking of treatments in decreasing order of similarity was unchanged for breeding bird detection rates: ULW®, felling/girdling, burn, and control (Table 4.2a). Control plots were always significantly less similar to the reference condition than all other treatments. Bird species detection rates that most influenced this rank varied somewhat among years. Northern cardinal (*Cardinalis cardinalis*) and tufted titmouse (*Baeolophus bicolor*) were the strongest contributors in all years, but tufted titmouse was more influential than northern cardinal in 1998 only (Table 4.2b). In 1996, pine warbler (*Dendroica pinus*) was the only other influential species. (Because of methodological problems associated with the spring 1996 sampling, we selected species whose detection rates were less affected by the time of the day [see Provencher et al. 1998]. We therefore caution the reader about the accuracy of similarity for 1996.) In 1997, the list of species that most contributed to similarity included great crested flycatcher (*Myiarchus crinitus*), pine warbler, and summer tanager (*Piranga rubra*). In 1998, the other influential species were red-cockaded woodpecker, red-headed woodpecker (*Melanerpes erythrocephalus*), and great crested flycatcher. Apart from northern cardinal and tufted titmouse, all other species are associated with open pine-grasslands (Ehrlich et al. 1988, Terres 1991, Wilson et al. 1995).

The common theme for detection rates in all years was a significant increase in similarity to the reference condition after any restoration technique was applied. Interestingly, species that associated more strongly with fire suppression (northern cardinal and tufted titmouse) (Ehrlich et al. 1988, Terres 1991) were the highest contributors to similarity (Table 4.2b). Another general, but weaker result was that similarity to the reference condition increased with the effectiveness of the hardwood reduction methods. We have shown that ULW® and felling/girdling treatments achieved greater hardwood reduction than a single growing season burn (Chapter 3). Restoration effects on similarity match this pattern, especially for ULW® application. However, we often observed that bird species detection rates were not significantly different among hardwood reduction techniques, despite there being large disparities in hardwood abundance. This trend applied to all species except northern cardinal in 1997 (Table 4.2b). We suggest that growing season fire may increase habitat suitability for species that prefer open longleaf pine forests to the point of compensating for greater hardwood presence. It is also plausible that some of these species tolerate a wide range of hardwood densities in the midstory, as has been shown for red-cockaded woodpeckers at EAFB (Hardesty, Gault, and Percival 1997).

Another aspect of breeding bird restoration ecology to consider is the differential effect of hardwood reduction techniques on the number of species. The number of species in control plots was significantly less similar to the reference plots than all other treatments in 1996 and 1997 (Table 4.2a). Felling/girdling plots showed the greatest similarity to reference plots in both years. In 1996, similarity was significantly greater in felling/girdling plots than in burn plots and ULW® plots. In 1997, felling/girdling, ULW®, and burn plots

Table 4.2a. Effects of hardwood reduction techniques on the similarity between restoration and reference plots for soil chemistry, arthropod variables, and breeding bird variables from spring 1996 to 1998 (post-treatment). Adjusted values from each treatment were ranked from highest to lowest. Higher rank indicates a greater similarity of treatment plots to reference plots for the examined variable. Inequality signs are only presented for significant contrasts. The "?" sign indicates an uncertain outcome for an untested contrast. Pre-treatment effects were factored out of these summary results, except for the number of breeding birds where no pre-treatment data are available. Similarity was measured by proportional similarity for all multivariate groups (i.e., all variables except number of bird species) and by the endpoint difference index for univariate variables (number of bird species). Soil chemistry was normalized with the Z-transformation and similarity values calculated with the endpoint difference index.

Variable	Highest ↔ Lowest	Highest ↔ Lowest	Highest ↔ Lowest
	Spring 1996	Spring 1997	Spring 1998
Soil chemistry	C = F = U = B†	F = B = C = U	...#
Herb-layer arthropods			
Family density	B = C = F = U	...§	...§
Morpho/species density	B = F = C = U	B > C ? U = F	...#
Soil/litter arthropod density	B = U = F = C	...‡	...‡
Breeding birds			
Detection rates	U = F = B > C	U = F = B > C	U = B = F > C
No. species	F > B = U > C	F = U = B > C	U = B = C = F

† Treatments: B = burn; C = control; F = felling/girdling; U = ULW® herbicide.
‡ Data not sampled.
§ Data not available in appropriate format.
# Data processing in progress.

Table 4.2b. Variables most likely to influence similarity patterns found significant for restoration effects on arthropod variables and breeding bird variables from spring 1996 to 1998 (post-treatment). Variables were ranked from contributing most to least weight to the similarity index value. Within variables, adjusted averages from each treatment were ranked from highest to lowest. Higher rank indicates a greater similarity of treatment plots to reference plots for the examined variable. Inequality signs are only presented for significant contrasts. The "?" sign indicates an uncertain outcome for an untested contrast. "(R)" indicates that the position of reference plots is untested. See Table 4.2a for statistical analysis.

Variable	Most Highest ←↕→ Lowest Least	Most Highest ←↕→ Lowest Least	Most Highest ←↕→ Lowest Least
	Spring 1996	Spring 1997	Spring 1998
Herb-layer arthropods			
Morpho/species density		*Sminthurus* sp. #1 (R) B = U = C = F*	
		Stilpon limitaris F > C = U = B (R)	
		Attalus sp. #2 C = U = F = B (R)	
		Pheidole floridana U = F = C = (R) B	
		Metcalfa pruinosa (R) B = C = F = U*	
		Leptothorax pergandei U = F > B = C (R)	
		Erythroneura spp. F (R) > B = U > C	
		Crematogaster ashmeadi C = U = F = B (R)	
Breeding birds			
Detection rates	Northern cardinal F = C = U = B (R)	Northern cardinal C ? F > B = U (R)	Tufted titmouse C = B = U = F (R)*
	Tufted titmouse F = C = B = U (R)	Tufted titmouse C = F = B = U (R)	Northern cardinal C = F = U = B (R)
	Pine warbler F (R) = C = B = U	Great crested flycatcher (R) F = C = B = U	Red-cockaded woodpecker (R) U = B = F = C*
		Pine warbler F = U = (R) B = C*	Red-headed woodpecker (R) F = B = U = C
		Summer tanager B = C = F (R) = U	Great crested flycatcher U = F = C = B (R)
No. species	(R) F > B = U > C	(R) F = B = U > C	

† Abbreviations: B= burn; C = control; F = felling; R = reference; U = ULW® herbicide.

* Significant experiment-wide restoration effect, although no contrasts were significant.

were not significantly different from one another (Table 4.2a). It is possible that abundant turkey oak resprouting in felling/girdling plots, a feature absent from herbicide-treated plots, may have created suitable habitat for additional species by increasing nesting opportunities and foraging substrate likely to harbor arthropod food items. Resprouting, therefore, may have increased the mean number of bird species to a level approaching the reference

condition (Table 4.2b). Alternatively, additional bird species may have been attracted to the park-like habitat of felling/girdling plots, which was more pronounced there than in burn and ULW® plots where dead standing trees remained. Results from 1998 do not support this hypothesis, however, because species richness in felling/girdling plots (15 species) exceeded that of reference plots (11.5 species), which were even more park-like. Bird species richness in ULW® plots increased from approximately 9.75 species in 1997 to 12.5 species in 1998 to match species richness found in reference plots. The median number of bird species was virtually unchanged at 10 species in burn plots from 1996 to 1998.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

We examined one of the most important themes of restoration ecology: how to use reference sites to measure restoration success while investigating restoration techniques (Palmer et al. 1997). We resolved two thorny problems in the process: the incorporation of reference sites into statistical tests of treatment effects without direct and inappropriate comparison to reference sites and the reduction of groups of variables into a simple metric of similarity. Because of these advances, we were able to process 55 data sets (including 17 pre-treatment data sets not shown in Tables 4.1 and 4.2, but implicit in the calculations) and to summarize the statistical outcomes of their analyses in only four tables without much loss of information.

The choice of a similarity index is important and should be motivated by the objectives of the study. Choosing one index of similarity among the many available and then selecting an appropriate variable transformation, however, involves some arbitrariness. Proportional similarity is widely used by ecologists, makes intuitive sense, takes advantage of quantitative data, and factors out site differences from calculations. Perhaps more importantly, traditional and commonly used indices of similarity, including proportional similarity, do not incorporate within-plot variance. For example, the difference in average relative abundance for the same species of plant may be large between two populations, but this difference should not be important if variances are sufficiently large to cause substantial overlaps between plot distributions. We found that the endpoint difference index solved this problem because it is based on the absolute value of the t statistic, which divides the difference between averages by their joint standard error. Moreover, the endpoint difference index, unlike other similarity indices, can be used on an ecologically related, but heterogeneous group of variables (e.g., soil chemical concentrations) after Z-transformation, which is widely used in statistics (Sokal and Rohlf 1981). We have not yet either studied the statistical properties or compared the sensitivity of endpoint difference to other accepted similarity indices. Although a few comparisons of proportion similarity and endpoint difference on the same data set agreed, further analyses are needed to test the effect of variance from groups of variables on these measures of similarity. A poor correlation between indices only in the presence of large plot-level variance would demonstrate the superiority of the endpoint difference index.

The main statistical limitation of the method proposed here was the repeated use of tests (ANCOVAs) among groups of similarity measures that may be ecologically interdependent. Usually, Bonferonni-type corrections (Jones 1984, Rice 1989) would compensate against the inflation of the significance probability for independent variables. However, because the different variable groups may be ecologically dependent (for example, insects feeding on

plants), a multivariate analysis is in order to arrive at conclusions about the whole system. We have not done a multivariate analysis here because it adds a layer of abstraction that could obscure findings. To compensate for this multivariate dependency, we have attempted to limit our conclusions within each group of variables (i.e., the rows of Tables 1 and 2) and not interpret our findings as a whole.

Tool for Monitoring

Using several assemblages of variables composed of single to large arrays, we proposed a method to test the effects of management to restore sandhills to their reference condition. Two products of the method are the quantitative identification of metrics potentially sensitive to changing degrees of hardwood encroachment (Tables 4.1b and 4.2b) and the endpoint difference index, a new metric of similarity that incorporates variability and works with heterogeneous, but ecologically related variables. We believe that this method could be adapted for use in experimental and quasi-experimental monitoring components of adaptive management programs at EAFB (Hardesty, Gordon, et al. 1997) and elsewhere. We would recommend, however, advanced ordination methods (Kenkel and Orlóci 1986, Clarke 1993) under non-experimental situations, such as those found with more descriptive sampling designs.

Treatment Efficacy

In previous reports (Provencher et al. 1997, 1998) and in Chapter 3 of this report, we suggested that the choice of one of the three treatments studied here depends primarily on the urgency to reduce hardwood midstory from a fire-suppressed to a maintenance condition. If a gradual approach is acceptable, growing season burning appears to be the most cost-effective restoration method. We have shown that growing season burning increased the similarity of plants, arthropods, and bird communities to those found in reference sites (Tables 4.1 and 4.2). Obviously, repeated burns will ensure long-term success, although we have demonstrated only short-term responses here.

On the other hand, managers may need an alternative to fire if they must rapidly modify the structure of the forest to meet certain objectives or if smoke management constraints prevent the use of prescribed burns. Reasons for quick action may be driven by single species management requirements and a perceived imminent threat to the ecosystem (e.g., an ecological threshold may soon be crossed). Mechanically or chemically eliminating the midstory will be expensive but effective solutions to these dilemmas (Chapter 3). Because the costs of felling operations and broadcast herbicide application are comparable, the choice between these options boils down to tolerance for resprouting and the side effects of herbicides on understory species. (In this study, we have not considered other mechanical methods of hardwood removal that involve machinery churning the soil.) Obviously, persistent smoke management problems will compel managers to use herbicides because, otherwise, resprouting without fire management is doomed to failure (Waldrop et al. 1992). If resprouting is incompatible with certain management goals even under fire management, herbicide may be needed. This situation may arise if, for example, fire can only be conducted during the dormant season, which may be less effective for oak control (Waldrop et al. 1992). We would not recommend ULW® as the herbicide of choice because we have shown that it negatively impacted groundcover plants directly, as well as in relation to

reference sites (Tables 4.1 and 4.2). Berish (1996) recommended the brushbullet version of hexazinone (Pronone®) as having less impact on groundcover plants. Herbicide injection by hypo-hatchet would also minimize impacts to the groundcover communities. The latter two herbicides, however, are more labor intensive to apply than ULW®. Finally, if resprouting does not hinder management objectives during fire intervals, we believe felling/girdling to be the less detrimental technique and cheaper than applying brushbullet or injecting herbicide. Moreover, bird species and arthropods appear to benefit from oak resprouting (Provencher et al. 1998, Tables 4.1 and 4.2).

5. RESTORATION OF NORTHWEST FLORIDA SANDHILLS THROUGH OFF-SITE SAND PINE HARVEST

ABSTRACT

Across much of the southeastern U.S., former longleaf pine (*Pinus palustris*)-dominated communities have been invaded by off-site pine species following logging of longleaf pine and fire suppression. In the southeastern portion of Eglin Air Force Base, longleaf pine was heavily logged prior to land acquisition by the Department of Defense in 1939-1940. Purposes of this study were to: a) document the impact of the Choctawhatchee variety of sand pine (*Pinus clausa* var. *immuginata*) (hereafter termed sand pine) removal from former longleaf pine-dominated sandhill sites on groundcover plant species richness and densities pre- and for 4 years post-removal; b) contrast results from removal plots to those from frequently-burned longleaf pine-dominated reference sandhills; c) test if sand pine seedlings colonized the periphery more rapidly than the core of the plots following tree removal; and d) measure the survivorship of planted containerized longleaf pine seedlings one year after planting. Vegetation was sampled prior to sand pine harvesting, which was done with heavy forestry machinery mounted on low-impact tires in 1995. Following harvest, vegetation was sampled in fall 1995. Plots were burned for fuel reduction and then planted with containerized longleaf pines. The number of plant species decreased from 30 to 26/400 m^2 from 1994 to 1995, but then increased beyond pre-removal levels to 46 species/400m^2 from 1995 to 1997, followed by a small reduction in 1998. The number of plant species was greater in reference plots compared to removal plots prior to 1997 only. New plant species in removal plots were predominantly ruderal and, therefore, not representative of the reference condition. Clonal shrubs and large trees were the only groundcover life forms whose densities continued to decline following sand pine harvest. Graminoids, non-legume forbs, legumes, non-clonal shrubs, woody vines, and small trees showed increases in density after harvest. Except for shrubs and woody vines, life form densities were greater in reference plots than in removal plots. Percent cover of graminoids, forbs, woody species, woody litter, and bare ground increased after the initial post-harvest drop. Predictably, fine litter cover declined erratically over time. Two to four years post harvest, the density of sand pine seedlings significantly decreased with the distance from the corner of the plots. No difference was observed one year post-treatment. On average, 78-81% of planted longleaf pine seedlings survived over the first two years in removal plots. Harvest of sand pine followed by fire and planting of longleaf pine may be an effective restoration approach in these sandhills invaded by sand pine.

INTRODUCTION

Human activities and uninformed management of natural systems can alter the dynamics and dominance of species. Various types of impacts not intended to cause site or community conversion can work synergistically to result in complete or near complete replacement of one community dominant by another (Denslow 1980, Robbins and Myers 1992). Once that dominance has shifted, often causing a change in resource availability, other species may follow. Examples of this type of change exist when both native (Robbins and Myers 1992) and non-native (Vitousek 1986, Gordon and Thomas 1997, Gordon 1998) species increase in the landscape.

This type of shift has occurred across much of the remnant longleaf pine (*Pinus palustris*)-dominated sandhill communities in the southeastern United States. Sandhills were once a common community type across much of the southeastern Coastal Plain (Frost 1993, Landers et al. 1995, Plunkett and Hall 1995). They are characterized by an open overstory of longleaf pine, with scattered hardwoods, such as turkey oak (*Quercus laevis*), and a diverse herbaceous understory (Myers 1990). Integral to the maintenance of community composition and structure are fires, thought to have occurred every 1-10 years. Lightning and early human activities established this fire regime, which was seasonally biased toward the growing season months of April-July (Robbins and Myers 1992). Frequent fires maintained the open structure of the community, top-killed many of the hardwoods which subsequently resprouted, stimulated flowering and seed production of several herbaceous species, created an ideal seedbed for recruitment of longleaf pine seedlings and herbaceous species, and excluded fire-intolerant species.

While indigenous people likely increased the fire frequency and may have ignited more fires outside the lightning season, larger scale alterations to the landscape have resulted from introduction of feral hogs that consume longleaf pine seeds and seedlings (Frost 1993), logging of longleaf pines, and fire suppression. One consequence of these activities is that more rapidly growing, less fire tolerant, and less desirable timber species have increased in abundance across much of the Southeast. This result is currently of conservation concern, because over 98% of the original sandhill habitat has been converted to other uses, and much of the remainder is highly degraded (Means and Grow 1985, Noss 1988, Frost 1993). Concern about habitat quality, threats to diversity, and viability of endangered species (e.g., red-cockaded woodpeckers [*Picoides borealis*]) has resulted in attempts to develop techniques for the restoration of degraded and some converted areas.

One location where past logging and fire suppression has resulted in large-scale shifts in community dominance is at Eglin Air Force Base (EAFB) in northwest Florida. Protected within the base are roughly 140,000 ha of sandhills and the third largest remaining population of red-cockaded woodpeckers (Department of Defense-Air Force 1993). This area may represent the largest remaining acreage of sandhills under public ownership (Johnson and Gjerstad 1998). However, in addition to the above-mentioned habitat disturbances, historical logging and military activities have resulted in extensive soil disturbance and landscape fragmentation. As a result, hardwoods (mainly turkey oak and sand live oak [*Q. geminata*]) dominate much of the area on the northwest portion of the base, and the Choctawhatchee variety of sand pine (*Pinus clausa* var. *immuginata*) (hereafter termed sand pine) dominates the southeastern portion (Department of Defense-Air Force 1993). This latter species is naturally dominant in scrub on barrier islands, in coastal areas, in mainland scrub sites, and in naturally fire-suppressed inland areas (Ward 1963, Parker and Hamrick 1996, Kindell et al. 1997). Individuals of all but the larger sizes of this species are killed by fire.

The natural encroachment by sand pine in fire-suppressed and logged sandhills can be relatively rapid. The Choctawhatchee variety of sand pine has non-serotinous cones (unlike the Ocala variety [*P. clausa* var. *clausa*]) and can produce viable seeds after merely five years of growth (Burns and Honkala 1990, Parker et al. 1997). Where sand pine has become dominant in EAFB sandhills, stands are characterized as having closed canopies of sand pine with sparse occurrences of older longleaf pine, dense subcanopies of sand pine, sand live

oak, and turkey oak, a sparse to moderate shrub layer, an herbaceous groundcover that consists of less than 5% cover of native perennial grasses, and up to 11% cover of lichens (Provencher et al. 1996, Kindell et al. 1997). Frequently-burned longleaf pine-dominated sandhills at EAFB typically have no lichens and >11% cover of grasses (Provencher et al. 1996, 1997). Verbal accounts from retired EAFB foresters and historical aerial photography from the 1940's suggest that longleaf pine was harvested from most EAFB southeastern sandhills during the 1930's. Sites were then managed for sand pine production by promoting the natural regeneration of this species through fire suppression (Spence 1962). These sand pine-dominated forests are not plantations. Currently, sand pine is primarily used for paper pulp.

Unlike the more conventional method of employing prescribed fire to restore sandhill communities dominated by midstory hardwoods (Myers 1990), restoration of sand pine-dominated sandhills requires more intensive methods. Prescribed fire usually will not carry through dense sand pine stands, and large individuals can escape scorching of the bole. Fires of the intensity necessary for killing sand pine are difficult to control. Mechanical removal of sand pine on a commercial scale, with fuel chipping or harvesting operations, is presently the best alternative. Mechanical methods open up the canopy with minimal damage to non-target tree species and generate revenues that can be used for future restoration efforts. The major disadvantages of mechanical removal are the intensive impact to understory species, soil compaction and rutting caused by heavy machinery (e.g., skidders, tractors, trucks, fuel chippers) (Greenberg et al. 1995), and the possible introduction of exotic plant species carried by contaminated machinery or other agents (Gordon and Thomas 1997, Greenberg et al. 1997). Moreover, these silvicultural activities, especially with site preparation, may lead to a shift of dominance from mature community species to native weedy species (Grelen 1962, Conde et al. 1983, Noss 1989).

Preventing recolonization of sand pine after its removal is also a challenge, especially where longleaf pine regeneration is attempted. Sand pine seedlings can originate from the seed bank or from seeds dispersed by larger trees remaining in restoration areas or at their periphery. Although fire would control the annually-produced seedlings, immediate fire management of these areas is challenging because fine fuel levels are usually too low to carry fire, and young, bolting longleaf pines may be vulnerable to scorching. Restoration needs of other species are unknown because little information about the persistence of understory species in these degraded sandhills exists.

The purpose of this study was to document the impact of mechanical removal of sand pine on groundcover plant species richness and densities at EAFB. Measurements were collected pre- and for 4 years post-removal. We contrasted groundcover values from sand pine removal plots with values from frequently-burned longleaf pine-dominated sandhills, where no recent stand thinning has occurred. Two management goals of this study were to: a) determine the rate of sand pine seedling colonization from the periphery of plots after harvesting, and b) measure the survivorship of planted longleaf pine seedlings for three consecutive years after planting.

METHODS

Site Description

EAFB occupies the southern portions of Walton, Okaloosa, and Santa Rosa Counties in the western Florida Panhandle (Fig. 5.1). The climate is temperate with mild winters and hot, humid summers. A more detailed description of weather, soils, and general botany is presented in Provencher et al. (1998) and Rodgers and Provencher (1999).

Removal Plots

We assessed vegetation composition, densities, and structure before and after mechanical removal of sand pine. Sites were selected in areas that had high densities of mature sand pine, but still had mature and seed-bearing longleaf pine present. These sand pine-dominated sandhills were classified as the "*Pinus palustris-Pinus clausa/Quercus laevis/Sporobolus junceus* Woodland Alliance" in The Nature Conservancy's *International Classification of Ecological Communities* (Weakley et al. 1998).

From the winter to summer 1995, removal operations were replicated in six 81-ha square plots located in the southeastern part of EAFB (Fig. 5.1: *S1* to *S6*). Except for the smallest trees, most sand pines were harvested. Pre-removal densities of sand pines >1.4 m high were 4.6 trees/200 m^2, whereas only 1.1 sand pine/200m^2 remained immediately after harvesting (unpublished data). Harvest areas, therefore, were clearcuts interspersed with longleaf pines and hardwoods. Plots were burned for fuel reduction from the fall 1995 to early winter 1996. Containerized longleaf pine seedlings were planted somewhat systematically shortly after the prescribed burns, thus resulting in variable densities throughout the entire plot. In plots *S1* and *S4* during 1996, hardwoods were felled by chainsaw in small areas where they were abundant.

Each 81-ha plot contained 32, 10 × 40-m sampling subplots. We arranged subplots in four linear arrays to examine the rate of establishment of sand pine and exotic species from the corners (periphery) to the center (core) of the square plots (Fig. 5.2). The linear arrays formed an "X", allowing for subplots to extend from periphery to core. This arrangement was more strategic than statistical, because these linear arrays are not blocks, as understood in experimental block design (Steel and Torrie 1980). In fact, one statistical block was formed by the four groups of four 10 × 40-m subplots at the periphery of the 81-ha (200-acre) plot, and the other block was comprised of the four groups of four subplots situated at the core (Fig. 5.2). In order to test sampling distance effects, subplots were spaced in 10- and 50-m sampling distances with two of each distance per periphery and two in the core between adjacent subplots. These distance tests are not presented here.

Reference Plots

A total of three pairs of 81-ha (200-acre) frequently-burned, longleaf pine-dominated sandhill plots were established to represent objective goals for the restoration of fire-suppressed plots (Fig. 5.1: *R1*, *R2*, and *R3*). Reference plots were not part of the restoration effort described above, but are a critical research component because they provide a benchmark for measurement of vegetation recovery in removal plots. Reference plots were

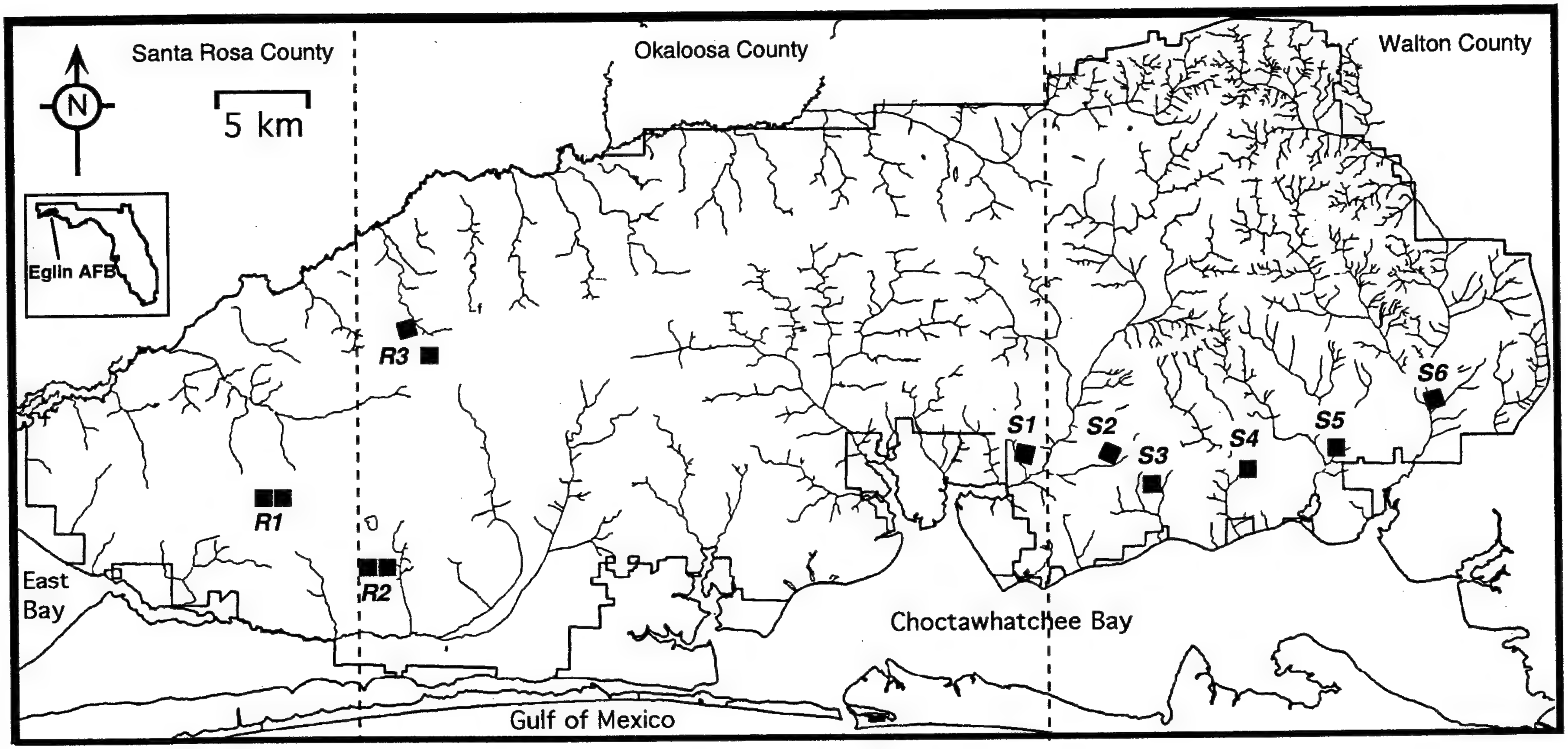

Fig. 5.1. Location of sand pine removal and reference plots on Eglin Air Force Base, Florida. Small squares represent 81-ha (200-acre) plots. Legend: *R1* – *R3* = reference blocks; *S1* – *S6* = sand pine removal plots.

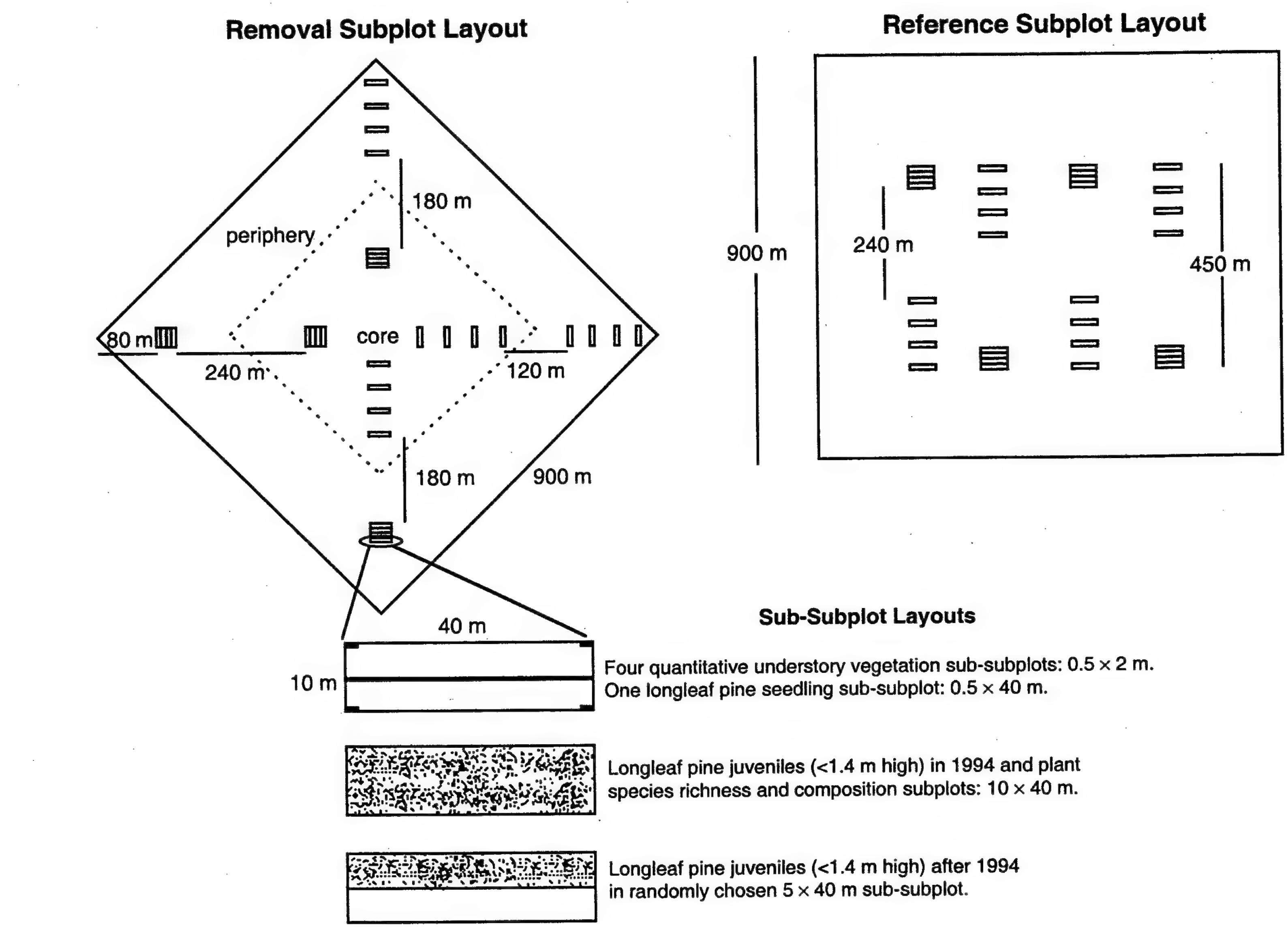

Fig. 5.2. Sample layout of 81-ha (200-acre) in one of six sand pine removal plots and in one of six reference plots. Spatial randomization of 10-m and 50-m subplot groups within core and periphery areas varies per sand pine removal plots.

chosen on the basis of the following criteria: a square area larger than 81 ha; uneven age distribution of longleaf pine; presence of old-growth longleaf pine; abundance of understory species interspersed with bare ground; reduced midstory; presence of active red-cockaded woodpecker clusters; and a history of frequent growing season fires (Myers 1990). We located only three blocks, each consisting of two 81-ha plots, satisfying these requirements. All other details on these plots are described in Provencher et al. (1998) and Rodgers and Provencher (1999).

Each reference plot contained the same number of subplots as the removal plots (Fig. 5.2). The positioning of the subplots within reference and removal plots differed, however. Groups of four subplots were placed in four parallel lines in the plot centers instead of the "X" design used in removal plots (Fig. 5.2). This arrangement reflected our desire to avoid potential edge effects on these sites. For the duration of the restoration study, reference plots are under a "let burn" management policy, which means that wildfires were usually not extinguished. All plots burned once and four plots burned at least twice during the study period.

Data Collection

We conducted pre-treatment data collection from August 1994 to October 1994, including densities of longleaf pine <1.4 m tall. Sand pine was harvested from January 1995 until July 1995. Late summer/fall post-treatment vegetation sampling was performed in August of 1995 to 1998. Longleaf pines <1.4 m tall were also sampled during the winter of the same years. Methodological details of vegetation and tree sampling, including definitions of life forms and cover groups, are presented in Chapters 3 and 4.

Statistical Analyses

We graphed the pre- and post-treatment average plot median, 25 and 75% quartiles, and minimum and maximum values of the number of plant species, vegetation life form density, and understory cover groups. (Three of six replicates closest to the median are contained within the 25 and 50% quartiles.) We chose to graph these statistics because they show the distribution of the data.

We tested by regression whether the density of sand pine seedlings (<1.4 m high) differed between the periphery and the core of the plots for 1994 to 1998 (Steel and Torrie 1980). We expressed the spatial position by the average distance of the subplots from the corner of the plots (distances obtained from Fig. 5.2). The density of seedlings from pre-removal sampling was used to show the initial condition. The null hypothesis was that of no difference between core and periphery, whereas the alternative was a greater density in the periphery because the density of dispersed seeds should decrease with distance from the plot's boundary.

We regressed the density of longleaf pine seedlings from 1996, 1997, and 1998 against the density of longleaf pines planted in 1995. Prior to 1997, nearly all counted seedlings were those that had been planted. Pre-treatment seedling density was negligible, and the fall 1996 seed crop did not contribute new seedlings in sampled sub-subplots (although a few zones of natural regeneration were observed). It should be noted that the pre-treatment density of mature longleaf pines was generally low (average = 0.32 longleaf pines/400 m^2) in

all plots. Importantly, the slope of the regression line should estimate seedling survival rate over one year.

RESULTS

Removal Effects on Groundcover Vegetation

The pre-removal median number of plant (trees and groundcover) species was slightly fewer than 30 species/400 m^2 in removal plots (Fig. 5.3). A few months after sand pine removal, species richness dropped to 25 species/400 m^2 and quartile distributions did not overlap with pre-treatment values. Two years post-removal, the number of species increased to pre-removal levels. There was substantial overlap between the 1994 and 1996 quartile distributions (Fig. 5.3). In 1997, the number of species jumped to 46 species/400 m^2, which was a 54% increase. Species richness then declined by 4% in 1998. This last sampling period coincided with a severe summer drought (National Oceanic and Atmospheric Administration 1998). A total of 56 plant species only found post-removal have been recorded from 1995 to 1998. Several of these post-removal species were native ruderals (i.e., weedy) and include coast sandspur (*Cenchrus incertus*), needle-leaf panic grass (*Dichanthelium aciculare*), poor Joe (*Diodia teres*), and sassafras (*Sassafras albidum*) (Provencher et al. 1998). In addition to the newly arrived ruderal species, the exotic Chinese tallow tree (*Sapium sebiferum*) and Chinaberry (*Melia azedarach*) have become established in two removal plots. Twenty-six species found in 1994 were not observed in 1998. None of these species were common in 1994. Forbs accounted for 57.7% (15) of these species, graminoids represented 11.5% (3), and the remainder were distributed among clonal shrubs, large trees, and non-clonal shrubs.

In reference plots, the median number of plant species increased slightly from 37 species/400 m^2 in 1994 to 44 species/400 m^2 in 1998 (Fig. 5.3), resulting in a 19% change. Quartile distributions overlapped among all years. Note that plant species richness was far more variable among reference than removal plots. Therefore, maximum species richness in reference plots always exceeded that of removal plots, although median richness was higher in removal plots than in reference plots in 1997 and 1998.

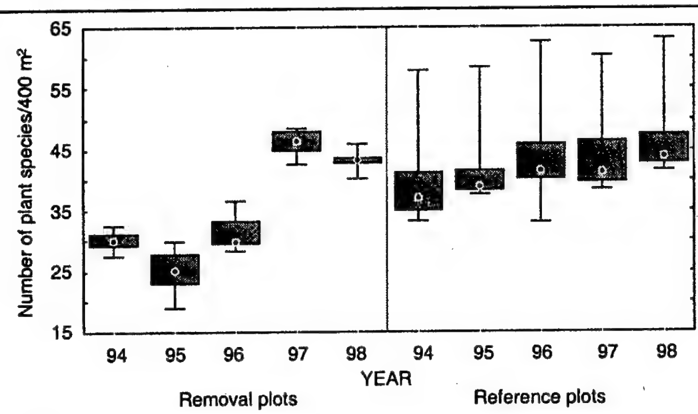

Fig. 5.3. Number of plant species per 10 × 40 m-plot in sand pine removal plots and reference plots during pre-removal (1994) and post-removal years (1995-1998). Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

Herbaceous Vegetation. The median density of graminoids decreased from 1 clump/m^2 in 1994 to 0.5 clump/m^2 in 1995, but then rapidly increased to, and stabilized at 4 clumps/m^2 in 1997 and 1998 (Fig. 5.4). The highest graminoid densities in removal plots were less than 50% of those of reference plots (8–10 clumps/m^2) (Fig. 5.4). In all years, median forb densities were under 3 stems/m^2 in removal plots, although an increasing trend was noticeable (Fig. 5.4). In reference plots, median forb density was at its lowest in 1994 (5 stems/m^2) and highest in 1997 (12 stems/m^2) (Fig. 5.4). The increase in density approximately paralleled that of removal plots, but the range of densities was much larger in reference plots and there was a notable decrease from 1997 to 1998 in reference plots. The general pattern of densities of legumes in removal and reference plots closely resembled that seen for forbs, except that median density of legumes did not exceed 3 stems/m^2 (Fig. 5.4).

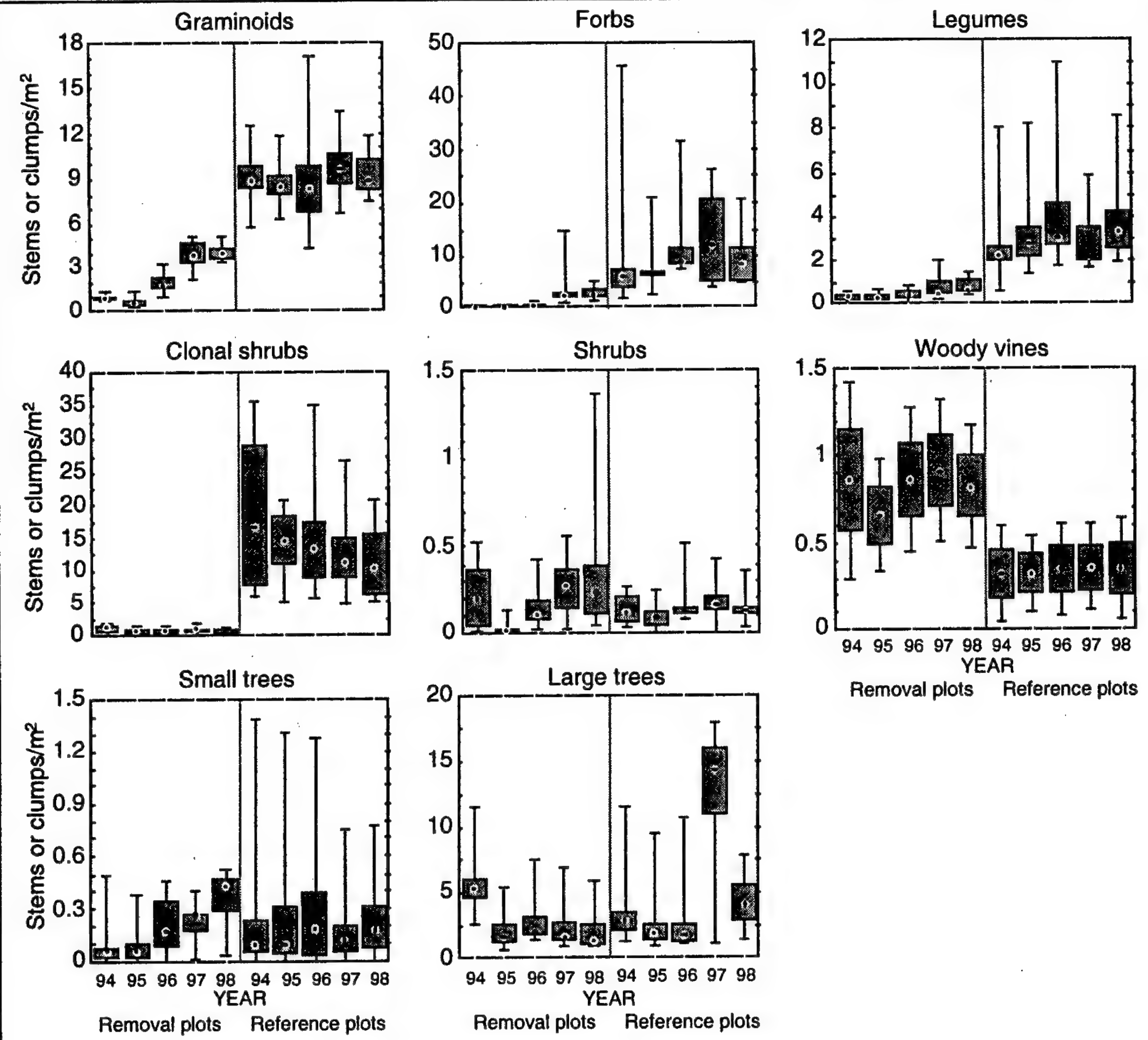

Fig. 5.4. Densities of different vegetation life forms in sand pine removal plots and reference plots during pre-removal (1994) and post-removal years (1995-1998). Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

Woody Vegetation. Clonal shrubs were the only life forms whose densities were initially low (<1.5 stems/m^2), and slowly declined following sand pine removal (Fig. 5.4). A trend towards declining clonal shrub densities in reference plots was evident from 1994 to 1998: 7 stems/m^2 were lost (Fig. 5.4) in 4 years. Median shrub density was 0.2 stems/m^2 in 1994, decreased to low values following sand pine removal, and then progressively regained pre-removal levels (Fig. 5.4). Densities of shrubs were approximately half as abundant and less variable in reference plots than in removal plots. Woody vines were always more abundant in removal plots (0.68–0.9 stems/m^2) than in reference plots (approximately 0.3 stems/m^2) (Fig. 5.4). Apart from the decline in density following removal, median densities were remarkably constant from 1994 to 1998 in both removal and reference plots. Median small tree density exponentially increased from 0.05 stems/m^2 in 1994 to 0.42 stems/m^2 in 1998. No such exponential increase was observed in reference plots (Fig. 5.4), where median density varied between 0.1 and 0.2 stems/m^2, with high variability among plots. Median large tree density (large tree species from the groundcover sample) was at its highest in 1994 (5.5 stems/m^2) in removal plots, but then decreased to approximately 1-2 stems/m^2 after sand pine removal with no evidence of recovery (Fig. 5.4). Except for a very high density of 14 stems/m^2 in 1997 attributable to longleaf pine seedlings from the mast crop of 1996, reference plot densities varied between 2.5 and 4 stems/m^2.

Understory Cover Groups

We present data for only seven cover groups (Fig. 5.5): all graminoids except wiregrass (*Aristida beyrichiana*) and pineywoods dropseed (*Sporobolus junceus*), forbs (including legumes and ferns), wiregrass and pineywoods dropseed, woody species, woody litter, bare ground, and fine litter. Lichen and cryptobiotic crust covers were not shown because of very low values in both removal and reference plots.

Herbaceous Cover. Graminoid and forb cover closely matched their respective life form density patterns (Fig. 5.4 vs. 5.5). Median graminoid cover increased in a sigmoid fashion from approximately 1% to 10% (we expressed proportion of cover as percent) in removal plots (Fig. 5.5). In reference plots, graminoid cover ranged from 11% in 1995 to 7.5% in 1996. Therefore, graminoid cover in the removal plots for the last two years was within the range of values observed in reference plots. Median forb cover generally increased in both removal and reference plots over time (Fig. 5.5). Forb cover increased from 1% to about 3% in removal plots, whereas cover increased from 3% to 10% in reference plots during the same period. Although present, wiregrass and pineywoods dropseed cover was practically undetected in removal plots in all years (<0.5%), whereas cover was <1% with highly variable median values in reference plots (Fig. 5.5).

Woody Species Cover. Median woody species cover decreased from 5% in 1994 to 3.5% in 1995, but gradually increased to 10% in 1998 (Fig. 5.5). In reference plots, median woody cover varied between 10% and 17% with no obvious trend. Median woody litter cover was approximately equal between removal plots in 1994 (5%) and reference plots in all years (Fig. 5.5). Woody litter cover in removal plots from 1995 to 1998, however, soared to 23% in 1995 after sand pine removal. Fuel reduction burns decreased woody litter cover to 13%, which then increased to 17-18% in 1997 and 1998.

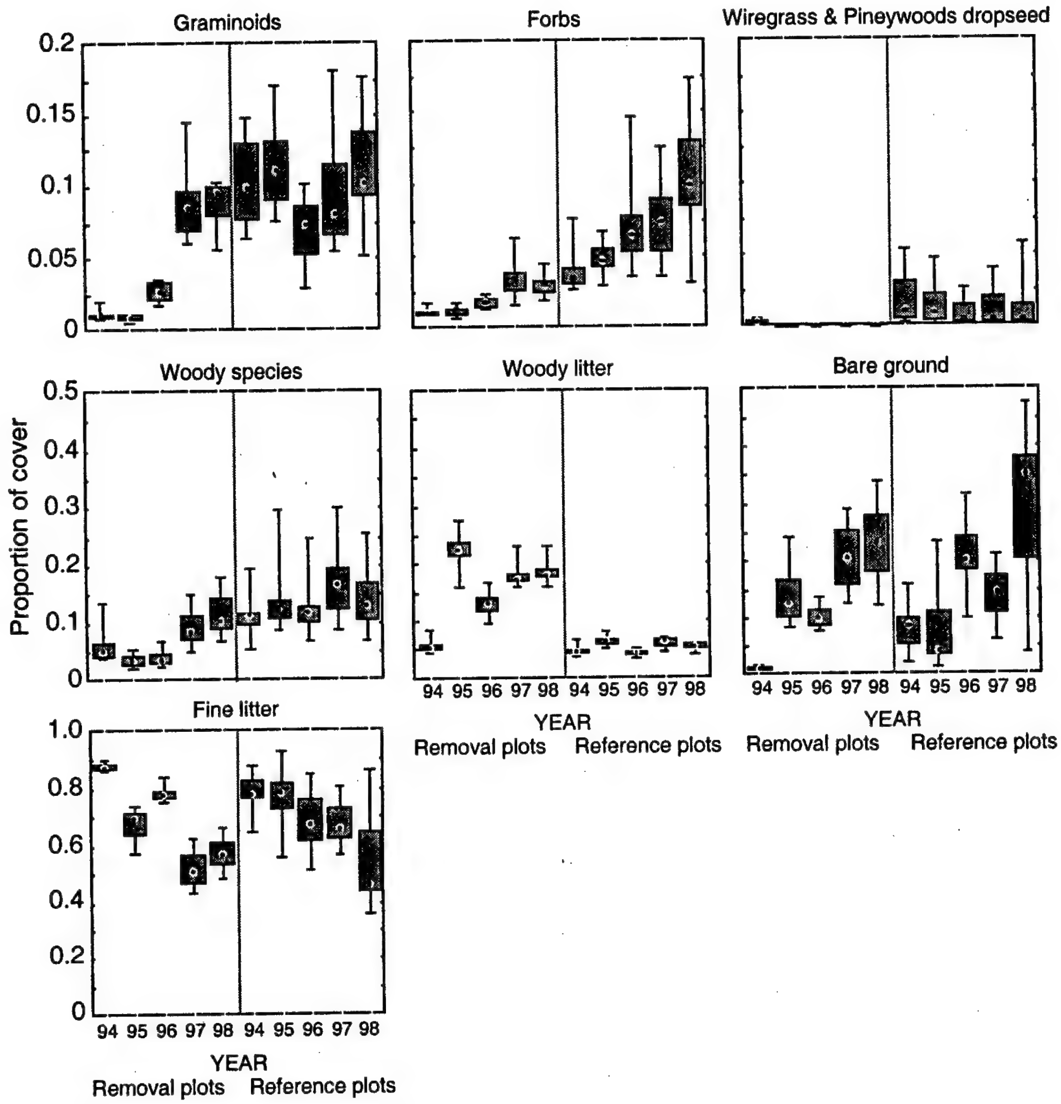

Fig. 5.5. Cover groups (expressed as a proportion) in sand pine removal plots and reference plots during pre-removal (1994) and post-removal years (1995-1998). Center of box represents the median, upper and lower edges of box are 25% and 75% quartiles, and error bars represent the minimum and maximum plot averages. Sample size = 6.

Bare Ground and Fine Litter. Bare ground cover was at its lowest in removal plots in 1994 (1%) and at its highest in reference plots in 1998 (36%) (Fig. 5.5). Fuel reduction burns in removal plots increased mineral soil cover after 1995 to >15%. Similarly, prescribed burns and wildfires in reference plots, which occurred after 1995, exposed mineral soil to >15% cover (Fig. 5.5). In removal plots, median fine litter cover, which varied between 71% and 89%, was approximately the mirror image of bare ground cover (Fig. 5.5). This negative association between fine litter and bare ground cover did not apply to reference plots, where median fine litter cover progressively decreased from 79% in 1994 to 46% in 1998.

Re-establishment of Sand Pine Seedlings

Prior to removal, densities of sand pine seedlings did not vary with distance from the corner of the plot in 1994 (2.3 stems/m^2; Fig. 5.6). Following sand pine removal, they became less abundant with distance (Fig. 5.6). The decrease of density with distance was not significant shortly after tree removal in 1995 (Fig. 5.6), but the decrease was significant from 1996 to 1998 (Fig. 5.6). In all years after tree removal, predicted sand pine seedling densities were between 0.12 and 0.18 stems/m^2 closest to the plot corner (approximate average distance of 108 m). Predicted seedling density was between 0.05 (1995) and 0.04 stems/m^2 (Fig. 5.6) 375 m from plot corners.

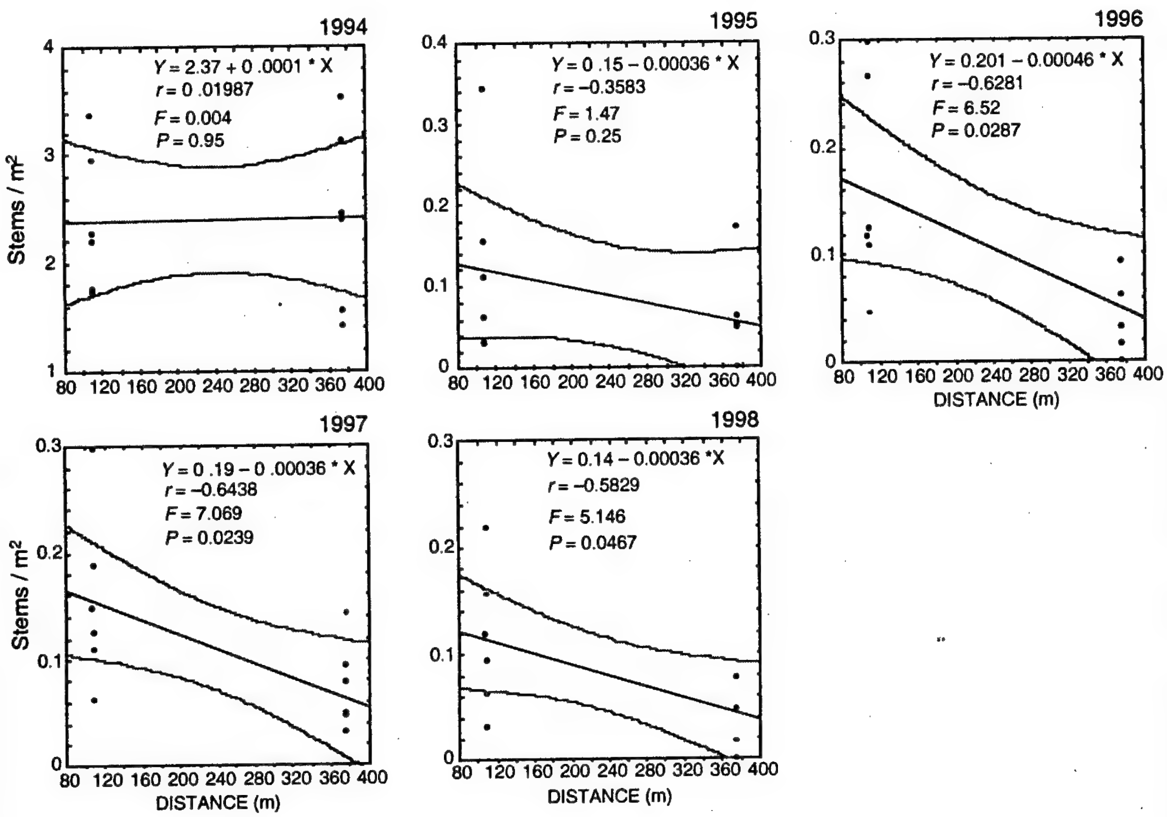

Fig. 5.6. Rates of sand pine seedling density with distance from harvested plot edge from 1994 to 1998. Upper and lower lines are 95% confidence intervals around the regression line. Each point is the average density per plot. Sample size = 12.

Artificial Regeneration of Longleaf Pine

The densities of planted longleaf pine juveniles were significantly correlated between 1995 and 1996 (correlation [r] = 0.814; Fig. 5.7) and between 1995 and 1997 (correlation = 0.796; Fig. 5.7). The slopes of the significant regressions in 1995-1996 and 1995-1997, respectively, were 0.78 and 0.81. Thus the first year survivorship was 78% and survivorship into the second year was 81%. Despite the high significance of the regression, many sample points were outside of the 95% confidence interval (Fig. 5.7). The qualitative shape of the

regression between densities in 1995 and 1998 completely departed from those of previous years, because we stopped differentiating between planted and naturally regenerated seedlings (Fig. 5.7). As a result, seedling density increased by an order of magnitude in many subplots. For example, longleaf pine seedling density increased from 26 planted individuals in 1995 to 206 total individuals in one subplot in 1998. The regression relationship was not significant, but recruitment was strongest in subplots with the lowest number of planted longleaf pine densities.

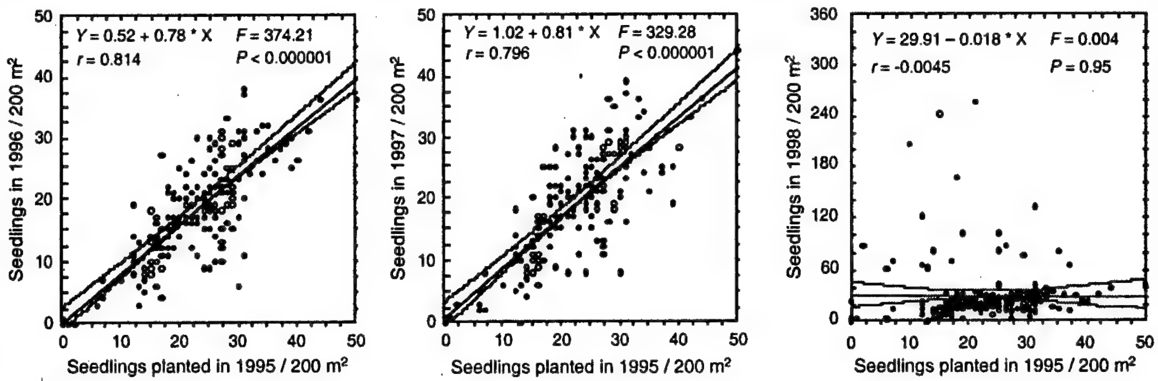

Fig. 5.7. Survivorship of planted, containerized longleaf pine seedlings from 1995 to 1996, 1995 to 1997, and 1995 to 1998 in sand pine removal plots. Sample size = 192.

DISCUSSION

Removal Effects on Groundcover Vegetation

The decrease followed by an increase exceeding pre-removal level for plant species richness (Fig. 5.3) was qualitatively similar to the one observed for graminoid density and cover, forb and legume density and cover, and for woody species cover (Figs. 5.4 and 5.5). Odum (1960) initially described this pattern of change in biomass/species diversity as applied to old field succession (also Golley 1965). The largest effect of removal operations on species richness was an increased detection of plant species, many ruderal, not seen during pre-removal sampling (Provencher et al. 1998). Earlier work in a chronosequence of thinned sandhills at EAFB similarly found that increased richness post-harvest may be attributed to an increase in native ruderals and do not represent desired components of the flora (Provencher et al. 1996). Furthermore, the newly arrived non-indigenous species threaten restoration efforts and justifies rapid post-harvest surveys to locate and eradicate them.

Modern silvicultural activities, which usually include site-preparation techniques to reduce competition, often eliminate key species, such as wiregrass, and may lead to a dominance of ruderal species (Grelen 1962, Conde et al. 1983, Noss 1989). Moore et al. (1982) reported short-term increases in species richness due to disturbance effects on understory communities in southeastern pinelands. Several species of grasses and forbs, many of which are ruderals, became established only after harvesting operations (Moore et al. 1982). Greenberg et al. (1995) observed an increase in herbaceous ruderal species in scrub plots where the Ocala variety of sand pine was removed by mechanical operations as compared to unharvested plots in the Ocala National Forest. Most native scrub species were

retained following the removal of the Ocala variety of sand pine, presumably because those species have evolved traits to survive and reproduce following catastrophic fires and hurricanes (Greenberg et al. 1995, Menges and Hawkes 1998). Greenberg et al. (1995) caution that their results may not apply when repeated harvesting (i.e., several rotations) occurs. In this study, mechanical site preparation other than slash reduction burns was excluded from forestry operations, which may explain why 85% (143 of 169 species) of pre-removal species, including wiregrass, survived this phase of restoration and benefited from subsequent burns (Provencher et al. 1998). This result is important because high survival of persisting species reduces the need for groundcover plant restoration (Seamon 1998), which is expensive and poorly developed. This positive result, however, does not imply that the composition of removal plots was restored compared to reference sites. Our results show that plant species composition, even as life form groups, did not resemble reference sites (Fig. 3). Earlier chronosequence work at EAFB (Provencher et al. 1996) suggested that at least 65 years of natural recovery is needed to return selectively thinned longleaf pine-dominated sandhills to a reference condition. A longer recovery time will be required for sand pine-invaded sites that contain only a few original longleaf pines.

Clonal shrubs, however, appear to be the exception to recovery among life forms because their numbers have continued to decline (Fig. 5.4). In our plots, this life form is primarily composed of dwarf huckleberry (*Gaylussacia dumosa*) and gopher apple, both of which are dominant in EAFB sandhills (Kindell et al. 1997, Rodgers and Provencher 1999). Dwarf huckleberry has persistently declined, whereas gopher apple has continued to increase since harvesting (Provencher et al. 1998). In the case of dwarf huckleberry, both damage to roots from forestry machinery (Provencher et al. 1996) and intense heat to the roots from fuel reduction burns may be the cause of this decline. In scrubby flatwoods of peninsular Florida, resprouting dwarf huckleberry density and cover peaked one year after burning (Abrahamson 1994). In the New Jersey Pine Barrens, black huckleberry (*Gaylussacia baccata*) roots have been shown to be negatively affected by intense fire, although the species quickly resprouted from rhizome after fire (Matlack et al. 1993). These results, therefore, suggest that mechanical damage to roots is the most likely cause of dwarf huckleberry decline in removal plots.

Increase of graminoids and forbs both in density and cover was evident (Figs. 5.4 and 5.5). An early low panic grass (*Dichanthelium* spp.) dominance phase was definitely observed in removal plots (Provencher et al. 1998). In northwest Florida, Grelen (1962) found that egg-leaf panic grass (*Dichanthelium ovale* [syn. *Panicum malocon* in Grelen {1962}]) and fall witchgrass (*Leptoloma cognatum*) were the first and most conspicuous perennial grasses two years after forest clearing. Egg-leaf panic grass was the dominant species in our removal plots. Seeds of low panic grass are typically large (0.8-3.2 mm long), have no structures for aerial dispersal, and, therefore, would be dispersed by animals. For example, low panic grass seeds are readily consumed by northern bobwhite quail (*Colinus virginanus*) (Grelen 1961). Seeds may also have been dormant in the seedbed. The forbs Florida spurge (*Euphorbia floridana)*, blazing stars (*Liatris* spp.), and pine barren pea (*Rhynchosia cytisoides*) also characterized this early revegetation phase (Provencher et al. 1998). Grelen (1962) found similar results for Florida spurge and blazing stars, but not for pine barren pea, in cleared forests of northwest Florida. Judging from other observations in recently disturbed sites in the southeastern U.S. (Grelen 1962, Moore et al. 1982, Campbell

1983, Conde et al. 1983, Provencher et al. 1996), we anticipated that broomsedge (*Andropogon virginicus*) and dog fennel (*Eupatorium compositifolium*) would dominate the biomass and, perhaps densities, of these plots within 3-5 years. Although broomsedge and dog fennel have increased appreciably from 1995, these species are not yet dominant.

Re-establishment of Sand Pine Seedlings

A welcomed result of removal operations was the reduction of sand pine <1.4 m tall by approximately an order of magnitude (Fig. 5.6). Fire and mechanical damage may have caused these effects by directly killing seedlings. Therefore, further seedling population growth from seed bank germination and dispersal is considered a major threat to the restoration of these degraded sites by EAFB managers (Department of Defense–Air Force 1993). We predicted that the number of sand pine seeds originating from adult trees remaining around the edges of removal plots would decrease with distance from plot boundaries. We confirmed that sand pine seedling densities were significantly greater in the periphery than in the core of removal plots by a factor of 2 to 3 in the second to fourth years post-removal (Fig. 5.6). Furthermore, we propose that the severe drought of 1998 may have killed seedlings because densities returned to the lower level observed in 1995. The vulnerability of sand pine seedlings to drought has been documented for the Ocala variety (Myers 1990, Parker et al. 1997).

The contribution of the seed bank may be, therefore, much less of a management problem than dispersal of seed from remaining trees. Larger removal areas appear to confer some restoration advantages because timber stand improvement operations, which are costly, might be avoided for the five years before less expensive prescribed burns can be used to control sand pine encroachment in young longleaf pine plantations. Five years is enough time for sand pine to dominate and start degrading the smaller restored sites and for fine fuels to accumulate to levels sufficient for carrying fire. Four years post-harvest, our field observations suggest that fine fuels were still too patchy to sustain fire.

Artificial Regeneration of Longleaf Pines

We estimated first-year and second-year survivorship rates at approximately 80% for EAFB (Fig. 5.7), which is high for Lakeland soil, similar to the 76% for bare-root plantings in disked, but more argillic soil reported by Farrar and White (1983). Survival rates of containerized longleaf pine seedlings are generally higher than bare-root survival within the same soil type, which justifies their two-fold higher costs (Seamon 1998). However, survivorship rates vary greatly with soil type and yearly rainfall (Seamon 1998).

Both the high survival reported here and natural recruitment in areas with lower initial densities of planted longleaf pine (Fig. 5.7) demonstrate that this species' dominance can be restored to these degraded sandhills at a relatively low cost and intensity of site preparation. With the exception of harvesting, which generated revenues, fuel reduction burning and localized hardwood felling were the only site preparation techniques used prior to planting. This minimal effort compares very favorably to the traditional practice of roller-drum chopping or applying herbicide to suppress herbaceous and hardwood competition.

MANAGEMENT IMPLICATIONS

Overall, silvicultural removal of sand pine with moderate soil disturbance followed by fire and longleaf pine planting appears to be a promising and cost-effective restoration approach in sand pine-invaded sandhills. We recommend that sand pine removal be conducted on sufficiently large scales to slow down recruitment of this species into cleared areas. Where significant numbers of native understory species have been lost to harvesting impacts or past degradation, more extensive restoration efforts than studied here will be necessary (e.g., Seamon 1998). We also recommend that monitoring of non-indigenous species be pursued in all sand pine harvesting areas of EAFB.

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